Eesti Maaülikool
Estonian University of Life Sciences

THE DETECTION OF LAND USE CHANGE AND ITS INTERACTIONS WITH BIOTA IN ESTONIAN RURAL LANDSCAPES

MAAKASUTUSMUUTUSTE MÕJU MAAPIIRKONDADE MAASTIKULISELE JA BIOLOOGILISELE MITMEKESISUSELE

ARE KAASIK

A Thesis
for applying for the degree of Doctor of Philosophy
in Environmental Protection

Väitekiri
Filosoofiadoktori kraadi taotlemiseks keskkonnakaitse erialal

Tartu 2014

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LIST OF ORIGINAL PUBLICATIONS

This thesis is a summary of the following papers, which are referred to by their Roman numerals in the text and included as appendices at the end of the thesis.

I Kaasik, Are; Sepp, Kalev; Raet, Janar; Kuusemets, Valdo 2011. Transformation of rural landscape in Hiiumaa since 1956: consequences to open and half-open semi-natural habitats. Ekológia (Bratislava) 30(2), 257 - 268.

II Leito, Aivar; Truu, Jaak; Õunsaar, Maris; Sepp, Kalev; **Kaasik, Are**; Ojaste, Ivar; Mägi, Eve 2008. The impact of agriculture on autumn staging Eurasian Cranes (*Grus grus*) in Estonia. Agricultural and Food Science 17(1), 53 - 62.

III Kaasik, A.; Raet, J.; Sepp, K.; Leito A. & Kuusemets, V. 2008. Land use changes on Hiiumaa Island (North-western Estonia) in the last fifty years. In: Ü. Mander, C.A. Brebbia, J.F. Martin-Duque (Eds), WIT Transactions on the Built Environment, Vol. 100, Geo-Environment and Landscape Evolution III: Third International Conference on Evolution, Monitoring, Simulation, Management and Remediation of the Geological Environment and Landscape, 16–18 June 2008, The New Forest, UK. WIT Press, Southampton, pp. 173 - 182.

IV Sepp, K.; Ivask, M.; **Kaasik, A.**; Mikk, M.; Peepson, A. 2005. Soil biota indicators for monitoring the Estonian agri-environmental programme. Agriculture, Ecosystems & Environment 108(3), 264 - 273.

V Roose, A.; Sepp, K.; Saluveer, E.; **Kaasik, A.**; Oja, T. 2007. Neighbourhood-defined approaches for integrating and designing landscape monitoring in Estonia. Landscape and Urban Planning 79, 177 - 189.

1.1. Author's contribution

I. The author is fully responsible for study design, data collection and analysis, and manuscript preparation.

II. The author is partly responsible for study design, data analysis and manuscript preparation.

III. The author is fully responsible for study design, data collection and analysis, and manuscript preparation.

IV. The author is partly responsible for data analysis and manuscript preparation.

V. The author is partly responsible for data analysis and manuscript preparation.

ABBREVIATIONS

AEP	agri-environmental program
AES	agri-environmental scheme
CAP	Common Agricultural Policy
CAPRI	Common Agricultural Policy Regionalised Impact
CORINE	Coordinated Information on the European Environment
CV	coefficient of variation
DPSIR	Driver-Pressure-State-Impact-Response
ED	edge density
ENIS	Estonian Nature Information System
EMR	Environmental Minimum Requirements
FAO	Food and Agriculture Organization of United Nations
GIS	geographical information system
LIM	Landscape Inventory and Monitoring - Swedish rural landscape monitoring program
LOWESS	locally weighted regression
MK	Mann-Kendall test
NILS	National Inventory of Landscapes in Sweden
OD	optical density
OECD	Organisation for Economic Co-operation and Development
PCA	principal component analysis
SD	standard deviation
SE	standard error
SQL	Structured Query Language
3Q program	Norwegian monitoring program of agricultural landscapes

1. INTRODUCTION

In scientific literature, the term landscape is usually not understood as "a static phase of a locality, but as a process continuing through time" (Palang 1998). In addition, change is considered to be an essential property of landscapes (Antrop 2003, Carranza *et al.* 2007), and landscape change has become one of the primary topics in landscape research (e.g., Forman and Gordon 1986, Mander and Palang 1994, Emmelin 1996, Jongman 1996, Forman 1997, Bastian 2002, Antrop 2003, Mander *et al.* 2004).

Different aspects of landscape change have already been studied in Estonia, e.g., Varep (1964) and Arold (1991, 2005) have resumed the knowledge about regarding regional differences and long-term dynamics of Estonian landscapes, Hellström (2002) has studied the development of the farming landscapes and settlements in Hiiumaa since the middle of the 19th century, Mander and Palang (1994, 1999) have revealed the main Estonian-wide trends and driving forces of land use dynamics during the 20th century. Some studies are oriented to perceptual landscape values (e.g., Palang *et al.* 2011), others predict the future of Estonian rural landscapes (Palang 1998, Palang *et al*. 2000, 2010) or use remote sensing for detecting changes in forests (Püssa *et al.* 2005, Liira *et al.* 2006, Peterson *et al.* 2006) or in agricultural land use (Peterson and Aunap 1998). Despite many studies, there remain scarce spatial studies on agricultural land use changes and biota relations, which are based on large-scale cartographical materials (1:10,000) and focused, in particular, on the impacts of the political and socio-economic reforms of the last half century.

In this work, landscapes are understood as complex, interrelated, dynamic, and hierarchic geo-systems (Arold 1991, 2005), which have formed as a reflection of natural and socio-economic processes. As the natural, socio-economic or political situation changes, land use is altered and leads to changes in the land use pattern, which, in turn, is the main spatially interpretable indicator of landscape change (Urban *et al.* 1987, Aunap 2007). In landscape ecology, biota is considered to be an integral part of a landscape as a geo-system, and, by now, several studies have proven the close correlation between landscape change and biodiversity (e.g., Burel *et al.* 2004, Grashof-Bokdam and Van Langevelde 2004, Hietala-Koivu *et al.* 2004, Bergman *et al.* 2004). Agricultural landscapes are important habitats for wild species, and changes in land use intensity and spatial patterns should have an influence on certain species as well as on the entire species composition of an area. Derived from this bilateral correlation, certain species could also be used for detecting changes in agricultural lands, i.e., for bioindication purposes. Realising the interrelated and dynamic nature of landscapes leads to a requirement to investigate the possibility of monitoring changes by analysing the changed land use patterns, incorporating suitable bioindicators at different hierarchic levels of landscapes and using complimentary data from several environmental monitoring sets.

The primary objectives of this thesis are the following:

- to analyse the structural changes in rural land use patterns that have occurred in Hiiumaa since 1956 and to determine the general tendencies and the main driving forces behind the changes (**I**, **III**).
- to analyse the correlation between the agricultural land use dynamics and biota from both ecological aspects, i.e., how the land use changes in recent decades have influenced the local numbers and distribution of autumn staging Eurasian Cranes (*Grus grus*) in Estonia (**II**), and, from the viewpoint of bioindication, i.e., whether certain soil bioindicators can be used for evaluating the effectiveness of agri-environmental (AE) measures as well as the level of human pressure at field level (**IV**).
- to analyse the spatial distribution of Estonian landscape monitoring and other complementary environmental monitoring networks to combine and integrate data from different monitoring sets and to achieve more coherent geographical coverage of monitoring data for the surveillance of agricultural landscapes (**V**).

This thesis consists of three main parts. After the introduction, the second chapter creates the theoretical background of the study, explaining, e.g., how the concept of landscape has developed and how the term landscape is understood in the specific context of this thesis. Additionally, this chapter provides a brief explanation of the "changing nature of landscapes, "landscape change-biota correlation" and the concept of landscape monitoring. The third chapter discusses the research preconditions, the scope, scientific hypotheses, objectives, methodologies and basic results of the case studies that formed the basis for writing papers **I-V**.

Paper **I** summarises the results of the follow-up study of rural landscapes in Hiiumaa Island (started in paper **III**) focusing, in particular, on the transformation of open and half-open semi-natural farmlands since 1956, as well as a special emphasis on the analysis of the impacts and possibilities of recent political and socio-economic reforms to rural land use change.

Paper **II** analyses the correlation between the agricultural land use change and biota, more specifically, the long-term dynamics of agricultural land use and the numbers of autumn staging Eurasian Cranes in Estonia to assess whether and how agricultural practice affects the local numbers and distribution of staging cranes.

Paper **III** analyses the transformation of the traditional diverse rural land use patterns in Hiiumaa to present more homogeneous landscapes. The detailed spatial study, which covered the main agricultural regions of the island with a total area of 267 km^2, was based on available large-scale aerial photos from 1956 to 1998 and on field studies in 2004/5. This extensive study is further developed, and the results are presented in paper **I**.

Paper **IV** analyses the correlation between the agricultural land use change and biota from the point of view of bioindication. The paper analyses the effects of agri-environmental program (AEP) measures to soil biota and the suitability of certain soil bioindicators (abundance, diversity, and ecological composition of earthworm communities as well as the hydrolytical activity of the microbial community) for monitoring human pressure, as well as the effects of AEP measures.

Paper **V** analyses how landscape features are covered by different monitoring data and how the current pattern of monitoring networks represents the landscape differences in Estonia. The nearest neighbourhood analysis of the landscape monitoring (three sets) and other complementary environmental monitoring networks (11 sets) was performed to explore the country's spatial coverage by stratified monitoring data, to combine and integrate data from different monitoring sets, and to examine the necessity of optimising or improving the developed monitoring concept and network for agricultural landscapes.

2. REVIEW OF LITERATURE

2.1. Landscape

Landscape has been attributed several different meanings and interpretations, and a single common understanding does not exist (**V**). Scientific literature provides comprehensive reviews on how the concept of landscape has developed and/or transformed throughout history (e.g., Sepp 1999, Klink *et al.* 2002, Antrop 2005, Kruse *et al.* 2010). Therefore, in addition to talking about landscape diversity as a modern topic in landscape studies, one can also talk about the diversity of meanings and understandings of the term itself in scientific, policy and belletristic papers.

The interpretation of the term, as well as the entire concept, has changed much as the integration of geological, biological and social sciences has resulted in a deeper understanding regarding the hierarchical and complex nature of the described phenomena and the factors and processes that are shaping these phenomena. In common language, landscape is simply a visual appearance of an area. Moreover, landscapes are meant as parts of the natural environment with a different nature, use and look (Arold 1991). This approach helps to distinguish e.g., natural and cultural, urban and rural, agricultural and industrial, diverse and homogeneous or scenic and damaged landscapes. In a more sophisticated sense, landscape is a territorial unit of certain size with several characteristic features (Kildema and Masing 1966).

In geosciences, landscape is usually understood as "a spatial formation of the Earth's certain surface area", in which geomorphological features are perceived as the main determinants of differences in water conditions, soils and vegetation (Arold 1991). Additionally, characteristic features, which are primarily the variations in relief forms with different genesis, shape, size and composition, provide the basis for classification of landscapes to typological and regional units at different hierarchic levels.

Ecological approaches emphasise the complex, interrelated, dynamic and hierarchical nature of landscapes. Landscapes are considered dynamic material systems, in which the biotic and abiotic components are interrelated with each other, in both their development and their spatial location (Arold 2005). By this concept, every landscape is a geo-complex or geo-system, in which a change in one component (land cover, vegetation, or the water regime, etc.) affects the other components and the entire complex (Arold 2005, **V**). Water plays an essential role in the interchange of energy and matter between the components of a certain geo-system, as well as between different geo-systems as territorial units (Arold 2005). In addition, mobile factors, such as animals, can be considered fluxes of matter and energy inside and between geo-systems. Humans can influence all the natural components of the geo-systems and, by now, most likely no landscapes can be found without direct or indirect human influences (Jones 1991). Although some authors (e.g., Arold 1991, 2005) consider human impact to be something outside of the natural geo-system, the majority of scientists tend to consider the anthropogenic factor as a consistent part of the landscape (e.g., Isachenko 1991, Bastian 2001, Pärn and Mander 2007).

An important change or development in the landscape concept is that a landscape is no longer understood as "a static phase of a locality, but as a process continuing through time" (Palang 1998). Moreover, change is considered an essential property of landscapes (Antrop 2003, Carranza *et al.* 2007). A landscape is generally considered to be four-dimensional, with the three spatial dimensions plus the time dimension. Therefore, there is no actual requirement to use the terms "landscape" and "landscape change" separately. It is generally recognised that present landscapes have formed as a result of natural and human steered processes that have occurred in the past. Similarly, the present landscapes have several future alternatives (Palang 1998, Palang *et al.* 2000, 2005, 2010), and "the choice between these alternatives depends on current policies, decisions, planning" (Sepp *et al.* 1999).

The European Landscape Convention defines a landscape as "an area, as perceived by people, whose character is the result of action and interaction of natural and/or human factors" (Council of Europe 2000). This definition is useful because it gives the landscape concept the fifth dimension, perception. According to the definition, it is important how people see and identify themselves in relation to a landscape, and this perception provides the opportunity to study not only the quantitative (objective) parameters of landscapes described by figures but also the qualitative (subjective) parameters, i.e., the cultural, historical, aesthetical, and emotional values of landscapes. This concept has deep roots in Estonia: the "perception aspect" of landscapes has already been emphasised in the 1970s (Eilart 1976). According to Palang (1998), a landscape consists of two interrelated parts, of which one is the objective, real and visible, and the other is the subjective, virtual and intuitive landscape.

A landscape can be defined simply as a characteristic structural pattern of land use/land cover or vegetation classes (Urban *et al.* 1987). This definition is suitable for all types of GIS analyses because this definition allows the landscape to be described simply by the geometry of its elements (size, shape and territorial arrangement) and by the given class or attribute values (Aunap 2007). In fact, many modern "landscape ecological studies focus upon the relationship between the spatial patterns and processes of flows of nutrients, matter and energy using landscape metrics as a tool to describe and quantify these changes" (Antrop and Van Eetvelde 2008).

The diversity in meanings and concepts does not suggest that there could be better (right) and worse (wrong) meanings and concepts. In every case, the chosen concept is largely dependent on the topic, the aims, the methodology that is used and the scope of the study. Thus, every scientist that focuses on landscapes should first explain how the term landscape is understood in the specific context of the study (Palang 1998). Because land use change is a key factor through which man reshapes landscapes (Lausch and Herzog 2002), in papers **I-III,** the term landscape is understood as a specific structural pattern of land use/land cover classes in a given time that has formed as a result of natural and socio-economic processes. The time dimension is introduced by comparing different land use patterns from different times, considering that changed land use patterns could reflect the processes that have been happening in a landscape and the driving forces behind the processes. Of course, it is impossible and even unnecessary to avoid subjective attitudes and assessments concerning these changes (the fifth subjective dimension), e.g., whether the changes are considered positive or negative, how the changes influence the aesthetic, historical or emotional values of landscapes, etc. For instance, although the ecological and cultural values of historical agricultural landscapes are generally recognised, these types of landscapes are rapidly vanishing in Estonia, in Europe and all over the world (Jongman 2002, Antrop 2005, Palang *et al.* 2006, Plieninger 2006). As stated by Sepp *et al.* (1999), "usually people idealise the old, but at the same time they shape a new landscape that differs drastically from that old ideal".

In developing the landscape monitoring program in Estonia (**V**) and in bioindication studies (**IV**) a more sophisticated approach to landscapes has been applied. In these papers, a landscape is considered a dynamic and interrelated geo-complex of biotic and abiotic components, or, as defined by Arold (2005), "a regional unit of material systems formed by the interaction of substances and processes within the geo-sphere". The studies that are summarised in this thesis focus on rural landscapes, as opposed to urban landscapes, which are used primarily for agricultural purposes.

2.2. Transformation of rural landscapes

Present landscapes have developed as a result of the dynamic interaction between natural and socio-economic processes. However, in recent centuries, human-induced changes have become even predominant in landscape changes throughout vast areas. These changes may involve all landscape components (Bastian *et al.* 2002), and, in this manner, entire landscapes as geo-systems (Arold 2005, **V**). Depending on the nature and intensity of human activity on a landscape, humans can easily affect matter and energy flows or even cause visible structural changes in landscapes, which can be described and analysed by changed land use/land cover patterns. Therefore, in recent decades, the analysis of various aspects of human influences on different landscapes has become an actual topic worldwide (Mander *et al.* 2004). According to Mander and Jongman (1998), the human impact in Europe differs from region to region, as determined by general land use purposes, e.g., in industrial areas, the landscapes are mainly influenced by construction and mining activities, as well as by fluxes in polluted air and water, and in northern and eastern Europe, the landscapes are also influenced by forestry. However, in general, because large parts of European landscapes have been in agricultural use for centuries, human impact is primarily determined by agriculture (Mander and Jongman 1998).

Several decades ago, it was realised that the rural landscapes in Europe were in a process of considerable transformation, and, when compared with natural factors, such as soil conditions, topography and climate, man-made decisions and actions had a more predominant role in this process (Meeus *et al.* 1990). By now, the change in rural landscape patterns can be observed almost anywhere in the world, and it is generally accepted that this process is primarily driven by changing socio-economic conditions and agrarian policies, including the improvement of agricultural practices and related land use changes. An increasing population and the growing need for food production in a globalised world has brought about similar developments in agricultural practices and similar tendencies in rural landscapes. According to Rabbinge and van Diepen (2000), a general shift from traditional extensive to modern intensive agriculture makes it possible to produce more on less land and with less labour. These agricultural changes have resulted in the intensification of land use in some areas and the marginalisation of other areas that are not suitable for intensive agriculture. According to Jongman (2002), primarily due to the intensification of agriculture, rural landscapes are simultaneously affected by the fragmentation of natural habitats and homogenisation of land use structures, which, in turn, could lead to the disappearance of regional differences and to as decline in traditional agricultural landscapes and their biodiversity.

Moreover, as shown by several studies, in recent decades, the environmental damage that has been caused by agriculture has increased significantly, and this increase is primarily related to the intensification of production (e.g., Stoate *et al.* 2001, Baldock *et al.* 2002). Although the indicators for agricultural intensification are difficult to define (Herzog *et al.* 2006), there is sound evidence of environmental and ecological damage of intensively managed areas. Thus, considering the complexity of negative impacts, the cost effectiveness of intensive agriculture compared with traditional extensive or modern forms of organic agriculture is highly disputable. In addition to adverse environmental and ecological impacts, such as a high potential for environmental or food pollution, degradation of soils, the disappearance of traditionally high-value landscapes and their biodiversity, intensive agriculture has several social, economic and cultural consequences as well: the loss of jobs, a decrease in rural populations and an increase in urban populations, the loss of cultural heritage related to traditional agriculture, etc. Intensive agriculture shapes uniform, more homogeneous, landscapes and, according to Muhar (1995), the agricultural landscape has increasingly lost its significance as a reflection of the cultural identity of a particular region (as mentioned by Bastian *et al.* 2002). This growing concern regarding rapidly vanishing traditional cultural landscapes and their diversity and identity values has also been expressed at the European Landscape Convention (Council of Europe 2000).

Although the perceptual, ecological and cultural values of traditional agricultural landscapes are generally recognised, these types of landscapes are rapidly disappearing all over the Europe (Jongman 2002, Antrop 2005, Palang *et al.* 2006, Plieninger 2006). It is generally accepted that the preservation of traditional agricultural landscapes requires continuous management, and this management cannot be achieved by passive or classical nature conservation methods, i.e., by the formation of protected areas. Because a landscape is a process, protecting traditional agricultural landscapes implies "sustaining the process, maintaining the land use practices that have created the landscapes" (Sepp 1999). According to Jones and Emmelin (1995), both abandonment and intensification lead to a change in its specific values, and this change is not always towards improvement. According to Dower (1998), the issue is not to freeze landscapes at some particular point in their long evolution but to manage the change in a way that sustains or even enriches the diversity and quality of landscapes.

According to Vos and Meekes (1999), in Europe, there remain positive perspectives for agricultural landscapes that are related to broader demands of the modern society from our landscapes. As mentioned by Bastian *et al.* (2002), in recent decades, there has been an increasing public demand for a healthy and scenic countryside as a part of the regional cultural heritage; many farmers move towards sustainability and multifunctionality, when the farmers gain profits from these changes, and the spectrum of different alternative management and farming styles, including organic, integrated, and recreational, has broadened considerably. In Europe, sustainable agricultural practices have been promoted as an agri-environmental scheme (AES), which has been designed to mitigate the negative effects of intensive agriculture, to support organic farming and to improve the environmental awareness of agricultural producers [Council Regulation (EC) No 834⁄2007]. Although the AESs notably vary between countries, their main objectives include reducing nutrient and pesticide emissions, restoring landscapes, protecting biodiversity and preventing rural depopulation (Kleijn and Sutherland 2003). In the United States, the negative effects of intensive agriculture have been diminished by a set-aside program, which was originally designed to control agricultural production, that now is primarily aimed at environmental benefits and the creation of wildlife habitats (Riffell *et al.* 2008). On the basis of voluntary contracts, the set-aside program provides a variety of financial incentives for landowners to convert croplands in environmentally sensitive areas to forests, grasslands and other forms of land cover (Riffell *et al.* 2008).

According to Bastian *et al.* (2002), the scope of investigations regarding landscape changes may vary from comprehensive surveys of landscape dynamics that cover a variety of phenomena and interactions to a limited number of landscape components that can reflect and/or indicate the conditions of the entire landscape as a geo-system. Therefore, the most promising approach to managing "a multitude of variable landscape features" is by focusing on "a few meaningful indicators" (Bastian *et al.* 2002). Because land use change is a key factor through which man reshapes landscapes (Lausch and Herzog 2002), many modern studies are focused on changes in spatial land use or land cover patterns that have been detected from available topographical materials, aerial images or remote sensing data, and use GIS and landscape metrics for describing and quantifying these changes (Haines-Young and Chopping 1996, Turner 1990). However, the calculation of landscape metrics could not be the sole aim. According to Li and Wu (2004), this calculation is appropriate only when it helps to improve the understanding and prediction of processes that occur in a landscape. The unnecessary or improper use of pattern indices in landscape analyses is caused by conceptual flaws, inherent limitations of indices or by landscape metrics that are often difficult to relate to processes that cause changes, as well as their consequences (Li and Wu 2004). The quantitative changes in land use patterns that are described by landscape metrics should influence qualitative parameters, such as the historical, aesthetical and ecological values of landscapes. There are several indices that describe the diversity, the size, the shape and the spatial arrangement of land use patches; however, these indices are difficult or, in some cases, even impossible to correlate to qualitative parameters (the perceptive values) of landscapes. Therefore, according to Uuemaa *et al.* (2009), in scientific literature, landscape metrics are used primarily in the context of biodiversity, habitat, and landscape change analyses, and "there are only a few articles on the relationships of landscape indices to social aspects and landscape perception, e.g., Franco *et al.* 2003, Palmer 2004, Lee *et al.* 2008".

Rural landscape changes in Estonia during the past century have been well studied and follow the general European trends (Vos and Klijn 2000, Van Eetvelde and Antrop 2003, Antrop 2005, Antrop and Van Eetvelde 2008). Existing studies have revealed the primary tendencies, such as the simplification and polarisation of land use patterns, a considerable increase in forests, a decrease in agricultural lands and the continuing decline of semi-natural grasslands, as well as the main socio-economic driving forces behind land use changes (Mander and Palang 1994, 1998, Palang *et al.* 1998). Some of these driving forces, such as land reforms in 1919, 1940, 1947 and 1989, deportations and collectivisation in the 1940s, and the formation of a military border zone along the coastline, are specific for Estonia. However, other more general factors that are related to the improvement of agricultural practices, urbanisation, etc., such as the concentration of agriculture, marginalisation, land amelioration, the use of bigger machines etc., have reshaped rural landscapes in many other countries and have brought about the loss of identity, coherence and diversity of agricultural landscapes (Ihse 1995, Jongman 2002, Luoto *et al.* 2003).

During the recent decades, Estonian rural land use has changed significantly as a result of recent political and socio-economic reforms (land reprivatisation since 1987, the reclamation of independence in 1991 and accession to the EU in 2004), and the change still continues. This change stimulates interest in studying the processes and in analysing their ecological and socio-economic consequences. From the 1990s, remote sensing has been used to detect changes in agricultural land use (Peterson and Aunap 1998). The need for remote data arose because of the collapse of the former state statistical data collection system during transition years. Simultaneously, remote sensing has also been used for analysing and detecting the changes in Estonian forests (Peterson *et al.* 1990, Peterson 1992, Püssa *et al.* 2005, Liira *et al.* 2006, Peterson *et al.* 2006). Recently, an airborne laser scanning technology, which is an alternative remote sensing technique, has been tested for landscape monitoring purposes. The new technology allows increased accuracy of measurements and extends the possibilities for 3D spatial analyses of landscapes or specific features, e.g., height measurements on woody vegetation (Müncher 2011).

Since regaining independence in 1991, the former strictly classified topographical materials have become available for research purposes. During recent decades, numerous large-scale vegetation maps, as well as repeated vegetation maps, have been compiled for selected areas of interest (e.g., Kalda 1991, Roosaluste 2010). The analysis of detailed aerial photos has helped to elucidate the changes in plant cover and land use in Estonian mire reserves and their neighbourhoods (Aaviksoo 1993). In addition, GIS analysis of historical maps, large-scale aerial photos and recent vegetation maps has enabled the study landscape history of calcareous (alvar) grasslands in western Estonia (Pärtel *et al.* 1999), etc. Some studies are oriented to future scenarios of Estonian rural landscapes (Palang 1998, Palang *et al.* 2000, 2010) because it has been generally recognised that present landscapes may have several future alternatives, which depend on today's choices, decisions, policies and planning (Sepp *et al.* 1999).

2.3. Agricultural land use changes and biota

Theoretical preconditions

In landscape ecology, biota are considered a consistent part of a landscape as an interrelated and dynamic geo-system, in which a change in one component (land cover, vegetation, or the water regime, etc.) affects the entire complex (Arold 2005, **V**). Humans can influence all the natural components of geo-systems and, in this way, have a direct or indirect impact on biota. In rural landscapes, the human impact on biota is primarily determined by agriculture, by both changed agricultural management practices (intensive *versus* extensive) and changed spatial land use complexity. Because agricultural landscapes are habitats for several species, changes in management intensity or in land use patterns should have an influence on certain species, as well as on the entire species composition of an area. Derived from this bilateral correlation, certain species could also be used for detecting changes in environmental and ecological states of agricultural lands, i.e., for bioindication purposes.

According to Mc Arthur and Wilson`s (1967) "island theory", it is generally accepted that, to survive, a species requires a minimum number of individuals. This theory suggests that several populations of species must be maintained at a minimum number and that an exchange between these populations is required. The metapopulation theory (Levins 1969), which is a spatially interpretable concept, states that, in fragmented landscapes, connectivity between biotope sites is vital for the survival of the subpopulations of the metapopulation (Merriam 1984). Therefore, to avoid the negative effects of isolation on biodiversity, it is important not only to preserve areas that are large enough for the survival of populations but also to maintain the possibilities for the exchange of species (Jongman *et al.* 2004). Several studies have demonstrated the positive effects of ecological linkages in fragmented landscapes for the survival of viable populations (Opdam 1990, 1991, Hanski 1998, 2004). Since the 1970s, the concept of ecological networks has been developed as a response to the fragmentation of natural areas (Sepp *et al.* 2002). This development has been simultaneous, and often independent, and, therefore, the concept is known by different names, including nature frame, network of compensative areas, system of landscape territorial stability, green network, green belts, greenways, etc. (Jongman and Kristiansen 1998). At present, the establishment of ecological networks at different spatial levels has become one of the most promising approaches through which ecological principles and biodiversity conservation requirements are integrated into spatial planning procedures and land use practices (Sepp *et al.* 2002).

Determinants of farmland biodiversity

According to Jongman (2002), rural landscapes are simultaneously affected by the fragmentation of natural habitats and by the homogenisation of land use structures, which, in turn, can lead to the disappearance of regional differences and to a decline in traditional agricultural landscapes and their biodiversity. The impact on biodiversity appears in two basic ways: first, large homogeneous agricultural areas could hinder the dispersal and migration of wild species in fragmented landscapes; second, the intensively managed homogeneous agricultural areas provide less habitats, and their biodiversity indicators are considerably lower compared with traditional heterogenic agricultural landscapes. Several studies have demonstrated that edges of ecosystems, called ecotones, are characterised by rapidly changing species compositions. In addition, this observation could be one explanation of the heterogeneous structure of landscapes, which is characterised by small fields and grasslands that are interspersed with non-crop habitats, such as woodlots, stone heaps, hedgerows, ditches, fences and field edges with natural vegetation, plays an important role in maintaining biodiversity in agricultural landscapes (Weibull *et al.* 2003, Purtauf *et al.* 2005, Marja 2007, Aavik and Liira 2010, Winqvist *et al.* 2011). Therefore, on higher (continental, national or regional) levels, the fragmentation of landscapes could reduce biodiversity, but on a lower (local) level, this fragmentation tends to increase the biodiversity. Different authors have emphasised that the interaction between landscapes and biota is scale dependent, hence, the need to explore the effects on landscapes or biota, as well as their mutual correlations on different spatial levels (Schweiger *et al.* 2005, Tscharntke *et al.* 2005, Herzog *et al.* 2006, Guerrero *et al.* 2011).

Although agricultural landscapes remain important habitats for wild species in Europe, providing habitats for more than half of all European species (European Environment Agency 2005), several current studies have shown that changes in agricultural land use intensity and landscape structure have resulted in a considerable loss of biodiversity (Chamberlain *et al.* 2000, Donald *et al.* 2001, Robinson and Sutherland 2002, Benton *et al.* 2003, Tscharntke *et al.* 2005, Geiger *et al.* 2010). Farmland biodiversity has decreased largely due to a sharp decline in semi-natural habitats that have developed as a result of long-term extensive agriculture (Ihse 1995, Luoto *et al.* 2003, Liira *et al.* 2008). Several studies have demonstrated high vascular plant species richness in long-term wooded meadows (Kull and Zobel 1991) and the highest numbers worldwide in the smaller spatial grain of temperate semi-natural grasslands (Wilson *et al.* 2012). For instance, in west Estonian calcareous wooded meadows, the management continuity is the most important determinant of plant diversity (Aavik *et al.* 2008). Another study shows that abandonment of traditionally managed wooded meadows can also negatively affect their epiphytic lichen communities, whereas the decrease in species richness and abundance is mainly related to increased canopy cover (Leppik *et al.* 2011).

Because it is impossible to reverse time and return to traditional land use practices, organic farming has been promoted as one solution to mitigate the negative effects of intensive agriculture on biodiversity. According to Winqvist *et al.* (2011), organic farming often increases species richness and abundance; however, the effectiveness of this method is often controversial and varies among taxa, as shown by separate case studies, e.g., Bengtsson *et al.* 2005, Hole *et al.* 2005, Fuller *et al.* 2005. In addition, when evaluating the effects of farming practices on biodiversity, the biogeographical aspects (Guerrero *et al.* 2011), as well as the effects of spatial landscape structure (Aavik and Liira 2010) and the neighbourhood aspects or the composition of the landscape surrounding arable lands (Cantero *et al.* 1999, Bengtsson *et al.* 2005), must be considered. For instance, as mentioned by Winqvist *et al.* (2011), primarily because of ignoring biogeographical distinctions, a recent European-wide study found no difference in bird species richness between organic and conventional fields (Geiger *et al.* 2010). Another extensive European study, which also included Estonia, shows that, at field and farm levels, the geographical location of study sites alone accounts for nearly one fifth of the total variation in farmland bird abundance, agricultural intensification alone accounts for only 4.3% of variation, and the largest share of variance (37.8%) is explained by the combined effect of both factors (Guerrero *et al.* 2011). However, because organic farming has been found to have variable effects on biodiversity, the question remains as to whether this method is the only effective way for increasing biodiversity and ecosystem services in agricultural landscapes (Bengtsson 2011).

The spatial heterogeneity of land cover or land use types is believed to increase the overall functional heterogeneity of agricultural landscapes, including food production, recreation, the maintenance of biodiversity and several other ecosystem services. Several studies have demonstrated that, compared with the effects of farming practices, the spatial landscape structure could have even greater importance for local biodiversity (Weibull *et al.* 2003, Grashof-Bokdam and Van Langevelde 2004, Purtauf *et al.* 2005, Hendrikx *et al.* 2007, Marja 2007, Aavik and Liira 2010). In general, higher farmland complexity tends to enhance biodiversity; however, similar to organic farming, this complexity cannot be beneficial for all species. For instance, an extensive pan-European study of farmland biodiversity demonstrates that organic farming and landscape complexity increased both species richness and the abundance of wild plants and breeding birds, whereas ground beetle species richness was unrelated to both farming practice and landscape complexity and the abundance even decreased with farmland fragmentation (Winqvist *et al.* 2011). In addition, it has been proposed that farmland species diversity and the potential for biological control could be the highest when organic farming is combined with heterogeneous landscapes; therefore, in homogenous landscapes, measures should be taken to increase landscape complexity (Winqvist *et al.* 2011). Fahrig *et al.* (2011) separately discusses the effects of compositional heterogeneity, which is the number and proportions of different land cover types, and configurational heterogeneity, which is the spatial arrangement of land cover types on maintaining and/or increasing biodiversity in agricultural landscapes.

Higher biodiversity in spatially heterogeneous or fragmented farmlands is primarily explained by increased habitat diversity, including edge habitats and microhabitats that are suitable for different taxa. Additionally, increased spatial heterogeneity could enhance the entire ecosystem complexity, e.g., increased species richness and abundance on lower trophic levels should increase the diversity on upper trophic levels as well. For instance, it has been observed that the spatially diverse fragmented landscape can increase feeding and breeding success for birds that depend on different habitats (Wilson *et al.* 1997). Separate studies have shown that the majority of farmland birds prefer a diverse landscape structure in which there are many different patches, such as hedges, stone heaps, vegetation of different height, etc. (Virkkala *et al.* 2004, Marja 2007). According to Uuemaa (2009), the correlation among landscape metrics, bird species diversity and habitat preferences has been studied extensively, whereas "of landscape configuration metrics, patch size has given the most important relationships with bird species richness", and this observation suggests that "fragmentation plays an important role for birds". Farmland biodiversity loss has been greatly linked to the intensification of agriculture, which has converted the former structurally diverse landscapes to large homogeneous fields and grasslands. Therefore, for every change in this landscape, the change could not be completely harmful for all species. There are always so-called "winner species", who can adjust, adapt or even take profit from the changed conditions. For instance, in Estonia, the flocks of migrating cranes and geese feed predominantly on large agricultural lands (cereal fields, cultivated grasslands and pastures) that have developed as a result of land reclamation for intensive farming. It has been suggested that their breeding success, as well as the species general state and abundance are, to a high degree, dependent on these agricultural resources (Leito *et al.* 2003a, 2003b, 2006, **II**).

The measures to maintain or improve farmland biodiversity

In the United States, a set-aside program has resulted in increasing population trends for most grassland birds, except for some species that are associate with open croplands (Riffell 2008). In EU states, the negative effects of intensive agriculture on local farmland biodiversity have been primarily relieved by AES. The effects of AES on biodiversity are, however, difficult to measure, and the existing studies are often controversial. This controversy is partly because, as stated by Kleijn and Sutherland (2003), "the application of the AES is highest in areas of extensive agriculture where biodiversity is still relatively high and lowest in intensively farmed areas where biodiversity is already low". In addition, more research is required to assess and to improve their actual influence on different taxonomic groups. Because the schemes could not be beneficial for all species, according to Winqvist *et al.* (2011), there is a conceptual choice: either the schemes are designed in a general way that benefits most taxa or the schemes should focus on certain taxa that are of conservation value (Elts and Lõhmus 2012) or that have high ecosystem service potential.

Several authors have emphasised the requirement for biodiversity indicators to evaluate the effectiveness of AES measures on farmland biodiversity (Kleijn and Sutherland 2003, **IV**, Knop *et al.* 2006). Despite the many articles on biodiversity indicators (OECD 2002, Büchs 2003), there is no such thing as a universal set of biodiversity indicators. When selecting indicators, it is important to adopt a hierarchical approach, which links the indicators to their respective level of analysis, such as field, farm or landscape (**IV**). Some indicators can have relevance only at specific scales of analysis, whereas others can be used at different spatial levels, e.g., the indicator "land use diversity" has significance at the level of "landscape", whereas the indicator "length of field boundaries" is more universal and applicable at different spatial levels (**IV**, Roose and Sepp 2010). Additionally, when evaluating the effects of specific measures on biodiversity, it is important to consider the time factor, i.e., how long the measures have been in use. Criticism towards AES is often because of ignoring the fact that the short-term application of AES in the framework of pilot studies cannot result in a significant improvement in farmland biodiversity (**IV**).

Bioindication

According to Bastian *et al.* (2002), "the human induced changes involve all landscape components, but to a different extent, whereby the most dramatic response can be expected from the biotic components (flora and fauna)". Certain species or groups of species react easily to changes in land use, management practices or environmental state of the farmland. Then, the species fulfil certain criteria (e.g., wide-spread, high abundance, restricted mobility, site specificity, easy sampling and taxonomy), which can be used for detecting changes in environmental or ecological states, as well as in the biodiversity state of an area. According to the stress-response model or to the Driver-Pressure-State-Impact-Response (DPSIR) chain, bioindicators can indicate the environmental or ecological state (S) of landscapes that have been affected by human pressure. Bioindicators are also used to assess the human impact (I) on certain functions, uses or values of the landscape. Bioindication is primarily used to detect human-induced changes in landscapes; however, according to Gerhardt (2002), bioindication can also be used to detect natural changes, e.g., natural successions with changed species compositions and environmental conditions. Bioindicators are useful when the indicated environmental factor is difficult to measure, e.g., pesticides, heavy metals, several other toxic interacting chemicals, etc. or when the factor is easy to measure but difficult to interpret (Gerhardt 2002), for instance, whether the observed changes have ecological significance, whether the enforcement of certain AES measures has given the expected results, etc.

Several bioindicators have been proposed for monitoring human pressure in agricultural landscapes: bumblebees (Sepp *et al.* 2004), earthworms (Haynes and Tregurtha 1999, Paoletti 1999, Büchs 2003, Schloter *et al.* 2003, **IV**, Ivask and Kuu 2008), soil nematodes (Tsiafouli *et al.* 2011), soil microbial communities (Büchs 2003, Schloter *et al.* 2003, **IV**, Truu *et al.* 2008), etc. Soil biota play a key role in the functioning of soils, and some of its parameters have considerable potential as early indicators of soil degradation or improvement (Haynes and Tregurtha 1999). For instance, earthworms are sensitive indicators of changes in soil health (Haynes and Tregurtha 1999, Paoletti 1999, Büchs 2003, Schloter *et al.* 2003, Ivask *et al.* 2006), and it has been suggested that some of its parameters can indicate the impact of human activities on the soil as well as the effects of certain AES measures (**IV**, Ivask and Kuu 2008). Paoletti (1999) suggests the use of biomass, species number, and ecological categories (epigaeic, endogaeic and anecique) as key parameters. According to Ivask *et al.* (2006, 2007) earthworm abundance and diversity in cultivated lands is generally lower than earthworm abundance and diversity found in undisturbed habitats. Agricultural activities, such as ploughing, various tillage operations, fertilising and the use of pesticides, strongly affect these animals (Paoletti 1999). Decreases in earthworm abundance can be directly attributed to injuries that are caused by cultivation practices or indirectly to habitat disruption and reduction in food supply, as well as high predation during and after tillage operations (Ivask *et al.* 2006, 2007, **IV**).

In addition, soil microbial community parameters have been widely used in monitoring soil quality (Büchs 2003, Schloter *et al.* 2003). The soil microbial community assures the degradation of organic residues and the biogeochemical cycling of minerals. According to Kandeler *et al.* (1999), soil microbial properties (microbial biomass, enzyme activities) enable the estimation of early changes in soil microbial processes and their impacts on the entire soil ecosystem (**IV**).

2.4. Landscape monitoring

Theoretical background and conventional approaches

Landscape monitoring is a rapidly developing approach in environmental science and management. As defined by Syrbe *et al.* (2007), "landscape monitoring is a regular, long-term surveillance of a landscape, aiming at early recognition, assessment and prediction of landscape change, and focusing on the effects of human impacts". Landscape monitoring can be perceived as one tool by which we can gain an overview of the state of the environment. This method provides essential data regarding the ways systems are changing and how rapidly. In addition, this method provides feedback regarding management so that we can assess and adjust our practices on a landscape (Roose 2005, **V**).

As summarised by Roose (2005), in several countries, special scientific research programs for landscape monitoring have been established (O'Neill *et al.* 1994, Ihse 1995, Herzog *et al.* 2001), and, in some countries, landscape monitoring programs have already been started (e.g., Bunce *et al.* 1993, Fuller and Brown 1994, Howard *et al.* 1995, Roots and Saare 1996, Ihse and Blom 1999, Groom and Reed 2001, Bailey and Herzog 2004). However, the key theoretical concepts and study objectives of landscape monitoring programs vary among countries. The first landscape monitoring programs primarily focused on land cover aspects (Bunce 1979). Over recent decades, landscape monitoring concepts have become more sophisticated, covering various landscape aspects from vegetation and biodiversity to the analysis of abiotic landscape components, such as water systems, soils and landscape structure, as well as anthropogenic and cultural aspects, such as landscape aesthetics and scenery (Bunce 1979, O'Neill *et al.* 1994, Ihse 1996, Seibel *et al.* 1997, Aaviksoo 1998, Mücher *et al.* 2000, Dramstad *et al.* 2001, Herzog *et al.* 2001, Bastian *et al.* 2002, Brandt *et al.* 2002, OECD 2002, Bailey and Herzog 2004, Groom 2004). Often, programs of landscape monitoring are policy driven (Groom and Reed 2001) or focus on specific values of landscapes. However, as stated by Haines-Young *et al.* (2003), landscape values change as societies and their natural capital change, and monitoring programs are adapted and developed accordingly.

In recent decades, certain efforts have been made to establish Europe-wide standardised monitoring programs. In addition to the most well-known, but most likely also the most criticised, CORINE Land Cover program (JRC-EEA 2005), there are several new promising programs, e.g., the Common Agricultural Policy Regionalised Impact (CAPRI) modelling system (Britz and Witzke 2008), which allows the estimation of the share of 30 different crops at 1 km^2 cell resolution for EU countries (Kempen *et al.* 2008); standardised surveillance and monitoring procedures for the European General Habitat Categories (Bunce *et al.* 2008), etc. Additionally, certain efforts have been made to integrate the datasets for the establishment of Europe-wide indicators for the state and diversity of the rural-agrarian landscapes (Paracchini *et al.* 2010). For testing the compatibility of national and Europe-wide monitoring programs, attempts have been made for converting data from national datasets to European standardised categories, e.g., the conversion of NILS data (from SQL database) into the European General Habitat Categories (Allard *et al.* 2010).

Many authors have emphasised that there are no readily available methodologies for landscape monitoring (O'Neill *et al.* 1994, Herzog *et al.* 2001, Groom 2004). The national landscape monitoring programs have been developed independently, and there are only a few reporting standards for agricultural landscapes, e.g., LIM in Sweden and 3Q in Norway (Blom and Ihse 2001, Fjellstad *et al.* 2001). There is, however, a set of basic principles for designing a monitoring program. For instance, when developing a landscape monitoring program, it is important to define the theoretical concept for monitoring, the objectives and specific objects to be monitored, as well as the criteria for selecting study areas, hierarchical levels, and optimal methods for data collection and analysis (**V**). In practice, every monitoring program is unique, primarily depending on geographical coverage, landscape features, the range of monitoring, available technology, and financial capacities. Although some aspects of landscapes, such as the structure of land cover or land use patterns, are often monitored through specifically designed landscape monitoring programs, several other landscape components, such as soil, habitat, and water, are monitored by independent programs (**V**).

Development of landscape monitoring in Estonia

Landscapes are changing continuously; these changes can be induced by natural processes or by human impacts. It is generally accepted that changing political or socio-economic conditions alter land use and sooner or later are reflected in changed land use patterns. Thus, the dynamics in land use structure can be used as an indicator of the socio-economic or political changes in society, as well as for the assessment of their impacts. In recent decades, Estonia has undergone drastic changes in agrarian polices. By the middle of the 1990s, it became evident that land reprivatisation and the formation of small-scale private farms resulted in extensive land abandonment, and this abandonment, in turn, could negatively affect the biodiversity and the aesthetic value of landscapes, as well as potentially raise the distribution of weed seeds and the danger of fire. Taking this general concern into account, the main objectives of the Estonian landscape monitoring concept were defined as follows:

- To determine the landscape structure.
- To follow landscape changes and to predict future trends on a national level.
- To provide statistics and an overview of the state of Estonia's landscapes.
- To obtain information that will enable the optimisation of the use of landscapes as a resource.
- To explain the correlations between landscape diversity indicators and other environmental characteristics.
- To compile a comprehensive reference list of Estonian landscape diversity.

In developing the landscape monitoring concept, several aspects were considered, including: available technology (GIS and spatial database tools, satellite and aerial images); the objectives and structure of existing Estonian monitoring programs; institutional and financial capacities; and the scientific principles of landscape ecology (Fig. 1 in **V**). In addition, the experiences from other countries were examined, e.g., "Landscape Monitoring and Assessment Research Plan" (O'Neill *et al.* 1994), "Countryside Survey 1990" (Bunce *et al.* 1993, Fuller and Brown 1994, Howard *et al.* 1995) and the LIM-project in Sweden (Blom and Ihse 2001), and some of these aspects were incorporated into the Estonian plan. In 1995, a draft concept of the Estonian landscape monitoring program (Sepp and Kaasik 1995) was prepared and presented to the Ministry of the Environment, and, by 1996, three landscape monitoring sub-programs, agricultural landscapes, coastal landscapes and land cover, were implemented (Table I in **V**).

A fundamental question in the development of any monitoring programs is the selection of monitoring variables and the design of the monitoring network. The monitoring network must be optimised in both spatial and temporal scales, aiming at the appropriate data density, data quality and efficient sampling strategies (Roose 2005, **V**). For statistical analyses, a random network design is the best option to exclude subjectivity and to provide landscape aspects the opportunity to be chosen by chance (Bunce *et al.* 1996, Brandt *et al.* 2002, Bailey and Herzog 2004). Alternatively, by a strategic approach, the country's territory is subdivided to more or less uniform regional units, and the monitoring sites are chosen to cover all of these uniform landscape regions. This approach is proposed to optimise the network design and to decrease the number of monitoring sites because large homogeneous areas require less sampling. However, for statistical analyses, it is always better to have more randomly selected sites, even if these sites are smaller in area.

The Estonia monitoring network for agricultural landscapes is designed by a strategic approach, which takes into account the spatial distribution of Estonian landscape regions. The concept, as well as the principles for network design, requires thorough rethinking. The application of an integrated landscape monitoring concept suggests that data for landscape analysis could be obtained not only from special landscape monitoring programs but also from other environmental monitoring programs. Paper **V** analyses the spatial distribution of different Estonian monitoring networks to combine and integrate data, as well as to optimise or improve the developed monitoring concept and network for agricultural landscapes.

3. CASE STUDIES

This chapter summarises the materials of five case studies that are presented in Papers **I-V**. Although the studies have been performed over a long period and have different objectives and methodologies, these studies all follow the same logics because these studies are all focused on rural land use changes, their interactions with biota and on monitoring the changes using spatial analyses and bioindication methods.

3.1. Transformation of rural landscapes in Hiiumaa since 1956 (I; III)

Estonian rural landscapes have undergone drastic changes during the past century and these changes continue. Although the general tendencies and the driving forces of land use change are well known (Mander and Palang 1994, 1999), there are few detailed spatial analyses regarding how traditional, diverse land use patterns have transformed to the current, more homogeneous landscape patterns, and there are only a few studies analysing the potential impacts of the recent political and socio-economic reforms (land reprivatisation since 1987, reclamation of independence in 1991 and accession to the EU in 2004) on rural landscapes and their biodiversity.

The hypotheses of the study were set as follows:

- On the basis of available decoded aerial images since 1956, it is possible to track the transformation of the former traditional diverse agricultural land use patterns to the present landscapes.
- The spatial land use change has an impact on farmland biodiversity.

Objectives

The main objectives of the study (**I**; **III**) were:

- To analyse the structural changes in rural land use patterns in Hiiumaa since 1956.
- To determine the general tendencies and the main driving forces behind the changes.
- To analyse the consequences of rural land use changes to open and half-open semi-natural land use types.
- To study possible correlations between the changed landscape patterns and the available bird count data for Hiiumaa Island.

Materials and Methods

The study area included two agricultural regions in Hiiumaa: Hellamaa (200 km^2), in the northeast, and Vanamõisa (~67 km^2), in the southern part of the island (Fig. I in **I**). In Hiiumaa, the share of agricultural land has drastically decreased from more than 65% in 1939 to less than 25% in 1992 (Mander and Palang 1999), and currently, most of the intensive agricultural land use in Hiiumaa is concentrated in these regions. Due to its relative isolation and poor preconditions for intensive agriculture (young, mostly stony and thin soils on marine sediments and limestone), the landscape changes in Hiiumaa have been slower, and the land use retained relatively traditional characteristics by the middle of the 1950s. Therefore, it was presumed that the first available aerial images from 1956 would reflect the diverse land use patterns that were developed during the first independence period of Estonia from 1918 to 1939.

The landscape analysis was based on decoded aerial photos (orthophoto maps) from 1956 and 1984. The large-scale orthophoto maps (1:10,000) were scanned, and the land use patterns of 1956 and 1984 were digitised in the GIS software MapInfo. The state of the present land use was identified by the digital Basic Map of Estonia (1:10,000), which was based on aerial images from 1998. By field studies in 2004/5, the actual land use situation was identified, i.e., the fields that remained in use, as well as long-term and short-term fallows. More than 30 combined land use/land cover types were distinguished. In 1956, a slightly more detailed land use classification was used, which allowed the differentiation of two additional agricultural land use types: pastures and fallowed fields. However, the classification has remained quite similar through the entire period. To define the former spread of semi-natural wooded meadows and pastures, as well as to analyse the successive transformations between the land use categories, the land use types were digitised as precisely as possible, e.g., for grasslands, pastures and mires, the existence of shrubs, trees or both was defined. The current distribution of preserved valuable open and half-open semi-natural habitats that were listed in the EU Habitats Directive was detected from the data of the Estonian Nature Information System (ENIS 2010). These data include the Fennoscandian wooded meadows and pastures (6530, 9070), the *Juniperus communis* formations on heaths or calcareous grasslands (5130), semi-natural grasslands (6210, 6270, 6280, 6410, 6430, 6510) and coastal meadows (1220, 1630). The land use patterns were analysed, and the first statistics and landscape metrics were calculated in the software MapInfo, MS

Excel and Fragstats. From landscape metrics, the simple conventional indices, such as the edge density, the number of patches, the mean patch size, the mean shape index, the Shannon's diversity index, etc., which were calculated for a set of patches that composed a class (land use type) or for an entire study area, were determined most appropriate to describe the changed land use patterns. In addition, the long-term monitoring data of the local numbers and distribution of autumn staging Eurasian Cranes (*Grus grus*) were used (Leito *et al.* 2003b, 2006).

Results

The detailed spatial analysis allowed the exploration of the changes and transformations in compositional heterogeneity and in configurational heterogeneity of rural landscape patterns, as distinguished by Fahrig *et al.* (2011). For instance, the spatial analysis helped to elucidate the origin of the larger reclaimed and ameliorated field systems and a significant simplification of field structures in our study areas. In addition, this analysis allowed the tracking of the gradual transformation of once wide-open and half-open grasslands to forests and other land use types.

The main results can be concluded as follows:

(1) The rural landscapes in Hiiumaa have changed considerably since 1956.

(2) By 1984, the traditional and extremely diverse land use pattern had simplified and polarised as a result of the intensification of farming, marginalisation, collectivisation and, particularly, land reclamation, which most likely had the greatest effect on this process. Because the major portion of reclamation works were performed between 1968 and 1979 (Fig. VIII in **III**), the greatest decline in total patchiness and edge density occurred between 1956 and 1984. By 1998, the general land use pattern remained approximately identical to that in 1984; only the majority of fields and grasslands were abandoned because the agricultural land use had attained the lowest level after the initiation of agrarian reforms and the collapse of collective farming (Fig. 1).

(3) The area of agricultural lands decreased because of grasslands by approximately 43%, and the area of forests increased by approximately 44% from 1956 to 1998 (Fig. IV in **I**). By 1998, more than half of the pastures and grasslands that were present in 1956 had been transformed into forests.

(4) By 1998, only approximately 3% of pastures and 12% of grasslands that were present in 1956 were maintained and classified as open grasslands (Fig. VI in **III**).
(5) The share of grasslands decreased significantly, particularly for natural types of grasslands, i.e., grasslands with shrubs and/or trees by our classification (Fig. V in **I**).
(6) The area of half-open land use types, such as grasslands and pastures that were sparsely covered with trees and shrubs, decreased significantly by more than 10 times from 1956 to 1984. The slight increase from 1984 to 1998 is most likely related to the abandonment of agricultural areas. In 2010, the valuable types of half-open habitats of wooded meadows and pastures have been preserved primarily in the coastal region of the Hellamaa study area, which has remained untouched from land reclamation (Fig. 2, Fig. VII in **I**).
(7) The ameliorated fields have been transformed primarily from grasslands and pastures (53%), former fields (30%), forests (12%) and shrubs (4%). The 163 larger ameliorated field-systems in 1998 have been transformed from more than 6000 land use patches that were present in 1956 (Fig. III in **I**).
(8) Because of the formation of small-scale farms since 1987 and land reprivatisation since 1992, many farmlands remained fallowed for short or longer periods (Fig VII in **III**). The general trend in land use between 1987 and 2004 was the abandonment and afforestation of farmland. However, the remarkable changes in land ownership and agricultural land use intensity since the end of the 1980s have not yet caused any significant changes in developed, simplified and polarised landscape patterns.
(9) The enforcement of subsidies for organic farming and Estonian cattle breeding since 2000, as well as a variety of EU agricultural subsidiary schemes since 2004, including special subsidies to support the management of semi-natural communities, provides the opportunity to direct current and future landscape change in a more sustainable way.
(10) The disappearance of traditional land use patterns and the sharp decline in half-open farmlands has decreased the spread of species-rich habitats. The accumulated spatial information of landscape change affords further challenges to study the effects of altered agricultural landscapes on biota.

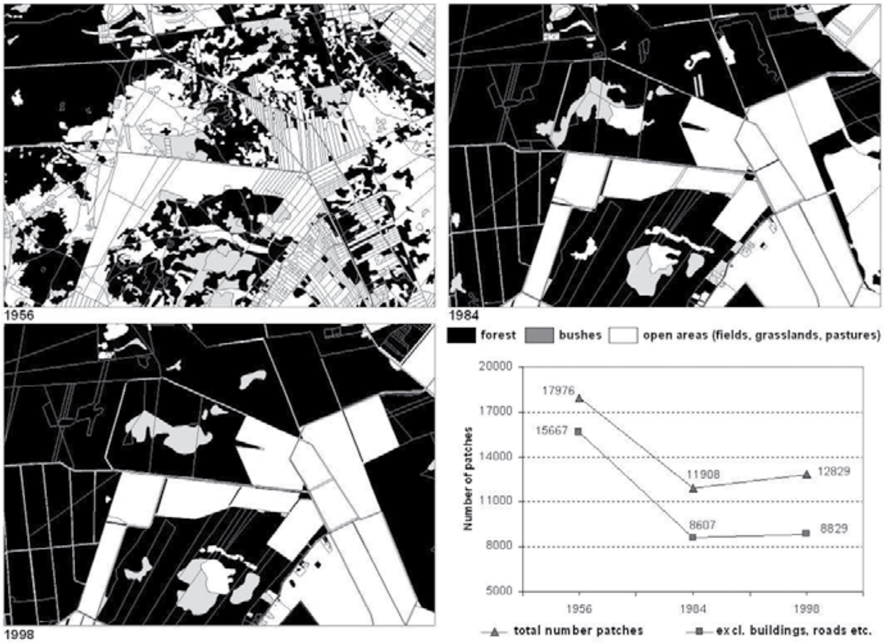

Figure 1. Changed patchiness and examples of changed land use patterns 1956-1998 (I).

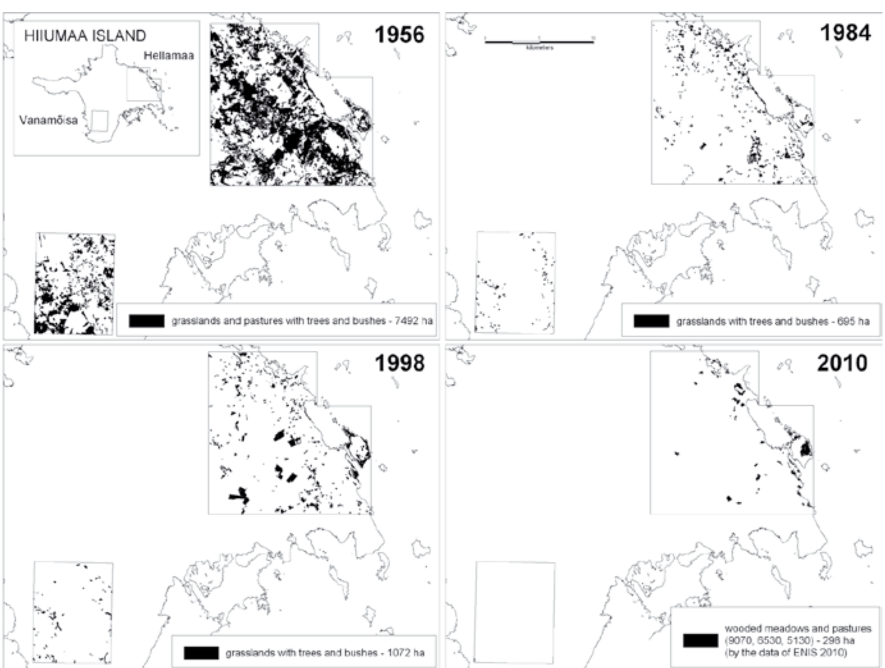

Figure 2. Disappearance of wooded grasslands in Hiiumaa 1956-2010 (**I**) (Habitat types according to the EU Habitats Directive).

Discussion

In Hiiumaa's agricultural regions, the changes during the Soviet period brought about the loss of traditional diverse land use patterns that had formed as a result of long-term farming practices. There were unrealistic expectations that the latest reforms of land reprivatisation, and the formation of small-scale private farms would help to restore the diverse land use pattern that was characteristic for the first independence period. In fact, a great portion of the privatised agricultural areas was left fallowed for short or longer periods and, as a general trend, land use changed towards the marginalisation and renaturalisation of agricultural areas (**I**, **III**, Raet *et al.* 2010). In addition, the remarkable revival of agricultural land use activity after the enforcement of the EU agricultural subsidiary schemes has not yet caused any significant changes in developed general landscape patterns. However, the incentives for organic farming and Estonian cattle breeding, as well as special subsidies that aimed to support the management of semi-natural communities, provide the opportunity to direct current and future landscape changes in a more sustainable way.

In 1956, the half-open land use types were widely spread in both study areas, which totalled 7492 ha (Fig. 2). It is now difficult to identify how much of these half-open lands were actually long-term wooded meadows or only recently abandoned farmlands that resulted from World War II, deportations and collectivisation. By 1984, primarily due to marginalisation, forestation and amelioration, less than 10% of such land use types were preserved (**I**). By 1998, the area of half-open lands slightly increased (1072 ha), but most likely due to the secondary afforestation of abandoned farmlands because the agricultural land use activity had most likely attained its lowest level by that time (Peterson and Aunap 1998). However, the general trend, which lead to the disappearance of extensive half-open farmlands, has continued until the present time (Fig. 2).

Fortunately, due to unfavourable conditions for intensive agriculture, the half-open farmlands, including the Fennoscandian wooded meadows and pastures (6530, 9070) and the *Juniperus communis* formations on heaths or calcareous grasslands (5130), remain apparent in many places (Fig. VII in **I;** ENIS 2010). When a comparison is made with intensive farmlands and economic aspects are disregarded, the scenic, biodiversity, cultural and potential tourist and recreational values of these landscapes remain considerably higher. Several studies have proven the high species richness of regularly managed long-term semi-natural temperate grasslands (Kull and Zobel 1991, Aavik *et al.* 2008, Leppik *et al.* 2011), which contain maxima global plant species (e.g., 89 species on 1 m^2) at smaller spatial grain (Wilson *et al.* 2012). Hence, the sharp decrease in half-open semi-natural land use types indicates a negative effect on farmland biodiversity.

The significantly changed land use intensity and landscape patterns of agricultural areas have influenced species composition, population numbers and the distribution of several species living, feeding or nesting in these landscapes. Although the impacts of intensive agriculture are generally related to biodiversity loss (Burel *et al.* 2004, Grashof-Bokdam, Van Langevelde 2004, Hietala-Koivu *et al.* 2004, Bergman *et al.* 2004), there are also exceptions to the rule. It has been observed that the developed simplified agricultural landscape with large homogeneous field systems has become an important feeding habitat for several migratory birds, e.g., geese and cranes. For instance, according to Alonso *et al.* (1987, 1994, 2003), some recent changes in the distribution of cranes on migratory routes and in wintering areas are clearly caused by changes in agricultural land use, depending on EU agricultural policy.

The main interest in the large-scale spatial land use data of the last 50 years was induced by the ambition to compare and combine these data with available bird census data from the same period. However, due to the long intervals of the spatial analysis (1956–1984–1998–2004–2010), it was impossible to determine statistically significant correlations between the fluctuating annual bird count numbers and the changed, but more inert, land use patterns. The developed homogeneous and spacious land use pattern seems to favour the general state and abundance of migrating and breeding crane assemblies; however, the actual land use of the agricultural lands, i.e., the area under certain field-crops and regular mowing of grassland, tends to have even greater effect on these birds. Recent studies by Ruskule *et al.* (2010 and 2012) have enlightened different aspects of the natural afforestation of abandoned agricultural lands, e.g., the renaturalisation of abandoned agricultural lands depends not only on time but also on several other factors, such as the size and shape of the field patches and their previous usage, the characteristics of soils, the land use of neighbouring areas, etc. For instance, in Hiiumaa, the developed general pattern of spacious field systems on thin and poor soils remained quite stable, even after long-term abandonment in the 1990s. For that reason, the following study (**II**) was focused on the potential correlations between the statistical changes in cropping areas and the count numbers of autumn staging cranes in Estonia.

3.2. The impact of agriculture on autumn staging Eurasian Cranes (Grus grus) in Estonia (II)

The recent political and socio-economic reforms in Estonia have resulted in significant changes in rural land use and agricultural production. The regain of independence in 1991, together with agrarian reforms and changed market conditions, brought about the extensive abandonment of farmlands and decreases in production (Csaki 2000). However, since 2004, the enforcement of the EU agricultural subsidiary schemes have resulted in a remarkable revival in agricultural land use and a broader spectrum of different alternative management and farming styles, including organic, integrated, and recreational farming (**I** and **III**, Raet *et al.* 2010).

The Estonian agricultural landscape is an important staging and feeding habitat for many migrating waterfowl and other species. For instance, the autumn staging Eurasian Cranes feed almost exclusively on larger cereal fields, mowed grasslands and pastures, and their local annual count numbers are considered to be dependent on these agricultural resources (Leito *et al.* 2006). In Estonia, the total number of staging cranes rose continuously during the 1960s and 1970s and stabilised in the period from the 1980s to 2000s. At the beginning of the 1960s, up to 5000 cranes were counted, and in the 1980s and 1990s, between 20,000 and 30,000 cranes were counted (Leito *et al.* 2006). It is estimated that approximately 10% of the European crane population stopover in Estonia during the autumn migration (Leito *et al.* 2006, **II**). In Europe, the breeding population of the Eurasian Crane is estimated to be 110,000 pairs (Prange 2003, BirdLife International 2004). The breeding populations have increased across most of Europe; e.g., in Estonia, the population has increased from approximately 300 pairs in 1970 to 6800 pairs in 2006 (Leito *et al.* 2003b, 2006). According to the hypothesis of this study the local differences in autumn staging crane numbers in Estonia depend on changes in agricultural land use in staging areas rather than on the size of the breeding or migrating populations.

Objectives

The main goal of this study (Paper II) was to analyse the long-term dynamics of agricultural land use and the numbers of autumn staging cranes in Estonia to assess whether and how agricultural practice affects the local numbers and distribution of cranes in staging areas.

Materials and Methods

This analysis was based on long-term monitoring data of staging cranes and on the statistical data of land use in Estonia. Autumn staging cranes have been monitored almost annually in the Matsalu area since 1961 and in Hiiumaa since 1982. Total crane counts throughout Estonia were conducted in the years 1983, 1994, 1999, 2000 and 2003 (Fig. IV in **II**). Censuses were performed using the standard method that was developed in Estonia (Keskpaik *et al.* 1986), and the data regarding the cropping area of different cereals, potato and hay in Hiiumaa and Läänemaa counties, as well as for all of Estonia, for the period from 1965–2005, were obtained from the archive of the Estonian Farmers' Union and from the Statistical Office of Estonia (2006). In Hiiumaa County (area = 1019 km^2), the cultivated areas of crops coincide with the feeding areas of cranes because the birds feed on all fields throughout the island. The crane staging population of the Matsalu region is spread over approximately 2000 km^2, which makes up approximately 84% of the territory of Läänemaa County (area = 2394 km^2). Consequently, the agricultural indicators for all of Läänemaa County are representative for the Matsalu crane population.

The coefficient of variation (CV) was calculated to demonstrate the temporal variation of crane numbers and cropping areas. The ordinations of cropping areas of different crops in Hiiumaa, Matsalu and in Estonia as a whole were analysed using principal component analysis (PCA). Spearman's rank order correlation coefficient (r_s) was used to examine the correlation between crane numbers and cropping areas during the same year in the Hiiumaa and Matsalu areas over the period from 1965–2004. The Mann-Kendall test (MK) was used to discover the presence of monotonic trends in the time series. Locally weighted regression (LOWESS) was used to illustrate trends in the time series data of crane numbers. Correlations and LOWESS were calculated using the computer program Statistica.

Results

(1) The study found changes in crane numbers and distribution in Hiiumaa and Matsalu regions and in all of Estonia. The numbers of autumn staging cranes have varied significantly in Hiiumaa and Matsalu regions, but in different ways.

(2) During the period from 1982–2005, the number of autumn staging cranes in Hiiumaa has varied by approximately four-fold between 960 in 1998 and 4230 in 1993 ($CV = 1.69$, $n = 21$ years), whereby four different periods in crane numbers can be distinguished: 1) relatively stable numbers at a low level in the 1980s; 2) a peak in numbers at the beginning of the 1990s; 3) a rapid decline in numbers until 1998, and 4) fluctuating numbers on a lower level from 1999–2005 (Fig. I in **II**).

(3) During the period from 1961–2005, the number of staging cranes in Matsalu has varied even more between 700 in 1960s and 21,500 in 1994 ($CV = 2.41$, $n = 30$ years), and there was a significant positive trend in crane numbers over the entire study period from 1961–2005 ($MK = 2.66$, $p < 0.01$, $n = 30$ years) (Fig. II in **II**).

(4) Based on five total counts between 1983–2003, the number of autumn staging cranes in Estonia has varied between 18,000 in 1997 and 30,000 in 1994 ($CV = 0.51$, $n = 5$ years). The total number of cranes has fluctuated without any visible trend (Fig. III in **II**).

(5) The main staging sites of autumn staging cranes in Estonia are in areas of large fields close to wetlands. The most important gathering area has been in western Estonia, including the islands of Hiiumaa and Saaremaa, where, in different years between 1983 and 2003, 72% and 87% ($n = 5$), respectively, of all cranes were counted (Fig. IV in **II**). The relative share of all staging cranes in Matsalu has been between 48–71% ($n = 5$) during the period from 1983–2003.

(6) There was a significant positive correlation between the number of cranes staging in Hiiumaa and in Matsalu ($r_s = 0.47$, $p < 0.05$, $n = 21$) and between Matsalu and Estonia as a whole ($r_s = 0.90$, $p < 0.05$, $n = 5$).

(7) During the period from 1965–2004, the total area of croplands in Estonia has varied significantly from 444,223 ha in 1980 to 259,248 ha in 2002 ($CV = 0.55$, $n = 40$) (Fig. Va in **II**). The dynamics of the total cropping area in Läänemaa, Hiiumaa and in all of Estonia has been quite similar (Fig. Vb in **II**).

(8) There was a steep decrease in cropping areas in the 1990s, both in Matsalu and particularly in Hiiumaa, where the cropping area

dropped almost to zero; however, over the entire study period from 1965–2004, there was no significant linear trend in the total area of field crops in Hiiumaa (MK = 0.32, $p > 0.05$, n = 49), Läänemaa (MK = 1.14, $p > 0.05$, n = 40) and in Estonia as a whole (MK = 0.47, $p > 0.05$, n = 40) (Fig. Va, b in **II**).

(9) The regional changes in the relative share of different field crops in Matsalu and Hiiumaa have been different compared with Estonia as a whole. In Matsalu and Hiiumaa, the relative share of different field crops was stable during the period from 1965–1990, but changed substantially in the period from 1991–1995, when the cropping area of all cereals together decreased (Fig VI in **II**).

(10) This study found a significant positive correlation between the number of staging cranes and the cropping areas of winter rye, winter wheat, summer wheat and all cereals combined and found a negative correlation between crane numbers and the cropping area of potato (Table 1). In Hiiumaa, the correlation was strongest with the cropping area of winter rye ($r_s = 0.58$, $p < 0.05$, n = 21) and winter wheat ($r_s = 0.58$, $p < 0.05$, n = 21). Additionally, in Matsalu, the correlation was strongest with the cropping area of all cereals combined ($r_s = 0.56$, $p < 0.05$, n = 28). No statistically significant correlation between crane numbers and the area of cultivated grasslands was found in either Matsalu or Hiiumaa.

Table 1. Significant Spearman's correlation coefficients (r_s) between the cropping area and the number of staging cranes in the Hiiumaa (n = 21 years) and Matsalu (n = 28 years) areas (**II**).

Locality	Crop	r_s	*p*-value
Hiiumaa	Winter rye	0.58	< 0.05
Hiiumaa	Winter wheat	0.58	< 0.05
Matsalu	Summer wheat	0.47	< 0.05
Matsalu	Potato	-0.41	< 0.05
Matsalu	All cereals together	0.56	< 0.05

Discussion

The Eurasian Crane breeding populations and the numbers of autumn staging cranes increased simultaneously through the 1980s. Then, the rise in staging crane numbers stopped, although the breeding population has continued to grow through the present day (Leito *et al.* 2003b, 2006). This difference is most likely related to agricultural land use changes. By the 1980s, the former traditional and extremely diverse land use pattern had simplified and polarised as a result of the intensification of farming, marginalisation, collectivisation and, in particular, land reclamation (**I, III**). Additionally, by that time, the coastal meadows, which were former (main) feeding and staging habitats for migratory birds, were largely overgrown with shrubs and reed beds (Ratas *et al.* 2010). Most likely, the newly cultivated lands, particularly large ameliorated field systems, have been the most important factor for the rise and development of autumn staging crane assemblies in Estonia (Leito *et al.* 2006). This study shows that the local differences in autumn staging crane numbers predominantly depend on changes in agricultural land use in staging areas, rather than on the size of the breeding or migrating populations. Therefore, crane numbers rapidly declined in Hiiumaa in the 1990s, whereas, in Matsalu, the crane numbers fluctuated or even increased since 1995, which most likely indicates the relocation of birds between the staging areas.

The population dynamics of migratory species is a complicated issue and certainly does not only depend on feeding success or migration, but on multiple other important factors, such as the availability of suitable habitats and food sources in breeding and wintering areas, nature conservation measures and hunting restrictions in wintering, breeding and staging areas, even on predicable climate change because wintering areas are located closer to breeding areas and because less energy and time is needed for migration, and also on agricultural policies over the entire distribution range, etc. (**II**).

The enforcement of AES and different subsidies for Estonian farmers has helped to enlarge the area of fields that favours the autumn staging of cranes. Because the schemes could not be beneficial for all species, according to Winqvist *et al.* (2011), there is a conceptual choice, either the schemes are designed in a general way that benefits most taxa, or the schemes should focus on certain taxa of conservation value (Elts and Lõhmus 2012). Several authors have emphasised the requirement to study and/or monitor the effectiveness of AES measures on farmland biodiversity (e.g., Kleijn and Sutherland 2003, Knop *et al.* 2006, **IV**) or even include AES in national landscape monitoring programs (**V**).

3.3. Soil biota indicators for monitoring the Estonian agri-environmental program (IV)

The implementation of agri-environmental programs (AEP) during the accession to the EU was a mandatory requirement for the Central and Eastern European countries. In Estonia, the development of the AEP began in 1997, and the implementation of the AEP in pilot areas was launched in 2001. The pilot AEP included a set of measures and activities that were intended for reducing nutrient and pesticide emissions, restoring landscapes and protecting biodiversity (Table I in **IV**). For instance, the aim of the "Environmentally Friendly Production Scheme" was to encourage the use of environmental planning by farmers and to reduce the risk of water pollution by nitrogen, while maintaining and increasing soil fertility. Farmers were required to have a Nutrient Management Plan and a Crop Rotation Plan. The total application of nitrogen (mineral fertiliser and manure) could not exceed an average of 170 kg ha^{-1} for the cultivated area, and the total application of nitrogen as mineral fertiliser could not exceed 100 kg ha^{-1} for the cultivated area. The crop rotation had to meet the following requirements: at least 5% of the cultivated area had to be under legumes or a mixture of legumes and grass species, and cereals could not be grown on the same field for more than three subsequent years. Elaboration of evaluation indicators for the AEP measures and testing them in pilot areas were also foreseen by the pilot project.

The hypotheses of the study were set as follows:

- Changes in agricultural land use and in the particular implementation of AEP measures have an effect on soil biota.
- Soil biota indicators can be used for monitoring the effects of the AEP measures, as well as the level of human pressure, on the field level.

Aims of the study

The main objectives of the study were as follows:

- To evaluate the effects of the pilot AEP measures on biodiversity and, in particular, on soil biota.
- To assess the suitability of soil bioindicators (abundance, diversity, and ecological composition of earthworm communities and the hydrolytical activity of the microbial community) for monitoring human pressure, as well as the effects of AEP measures.

Materials and methods

Two pilot areas of the AEP with contrasting natural conditions and management intensity (Palamuse in Jõgevamaa, representing intensive agriculture, and Kihelkonna-Lümanda in Saaremaa, representing extensive agriculture) and two reference areas, where the AEP was not implemented (Mustjala and Saare municipalities in the same counties), were selected (Fig. I and Table III in **IV**).

The agriculture of the Saaremaa study area was already extensive during the Soviet period (before 1991), and the use of agrochemicals was much lower than the Estonian average. Since the end of the Soviet era, the use of agrochemicals has further decreased. Between 2001 and 2003, most of the farmers in the Saaremaa pilot area used little or no mineral fertilisers, and pesticide use had dropped to almost zero. None of the monitored farms were using any pesticides. The average size of fields was 2 ha. The main soil type, according to FAO classification (FAO-UNESCO 1994), is pebble rendzinas (*Rendzic Leptosols*). In Palamuse, which is a typical area of intensive agriculture, the use of fertilisers was much higher (up to 170 kg ha^{-1} of NPK). Additionally, pesticides were used primarily for cereal cultivation, with average quantities between 0.8 (mixed crop) and 3.8 l ha^{-1} (oat). The average field size in Palamuse was 4.8 ha. Typical brown soils (*Calcaric Cambisols*) and pseudopodzolic (*Podzoluvisols*) soils are dominant. In both pilot areas, 10 farms that had joined the AEP and had implemented the measures of the "Environmentally Friendly Production Scheme" were selected, and in each farm, a cereal field was selected for investigation. For both pilot areas, reference areas (Mustjala for Saaremad and Saare for Palamuse) were selected. Additionally, in both areas, five farms that did not participate in the AEP were monitored. In these farms, fields of similar size and with the same crops as the AEP study farms were chosen.

In both pilot test areas of the AEP, soil biotic indicators were measured in 2001 and 2002. Monitoring parameters of the earthworm populations were as follows: the abundance of earthworms, number of species, biomass per m^2, mean fresh body biomass, ecological composition of community, and dominancy. Additionally, the hydrolytical activities of microbial communities were determined. Soil moisture content and acidity, which are the most important limiting factors for earthworms, were also measured. In all study fields, five randomly selected soil blocks of 50 cm x 50 cm x 40 cm in the centre of the field were studied by the hand-sorting method (Satchell 1969, Meyer 1996). Samples were washed and weighed, and species were identified according to Graff (1953) and Timm (1999). The mean number of individuals per m^2 of soil surface and standard error (SE) and the ecological composition and dominancy of communities were calculated. In all composite soil samples, the gravimetric moisture content (at 105 °C), pH (KCl), total N (Kjeldahl) and organic matter content (at 360 °C) were measured. The total activity of the microbial community was measured using the fluorescein diacetate method, which estimates the activity of dehydrogenase enzymes in a composite sample (Schnürer and Rosswall 1982). All data were analysed using the dispersion analysis of Kruskall-Wallis.

Environmental Minimum Requirements (EMR), which were baseline values, were identified for each monitoring parameter in both study area and were compared with their actual values. EMR values were calculated from the national environmental monitoring data from the period from 1995-2001: EMR = mean value – standard deviation (SD), n = 60 (Table III in **IV**).

Results

(1) There were statistically significant differences in some soil characteristics (nitrogen concentration, soil pH). Saaremaa soil pH was 7.2 ± 0.2; Palamuse soil pH was 5.9 ± 0.2. Organic matter content and total nitrogen concentration were also higher in Saaremaa soils compared with Palamuse soils (organic matter content 7.5 ± 2.0 and 2.8 ± 0.3%, total nitrogen content 0.47 ± 0.14 and 0.14 ± 0.02%, respectively).

(2) Differences in abundance and number of earthworm species between AEP pilot areas and their reference areas (Lümanda-Kihelkonna and Mustjala; Palamuse – Saare) did not exist. In both Saaremaa and Palamuse test areas, six earthworm species were found. Only species that were tolerant to agricultural activities (one epigeic, three endogeic, two anecic) were discovered, and more sensitive species were not found. The dominant species in communities was *Aporrectodea caliginosa* (81 and 89%, respectively of all individuals).

(3) In the Palamuse area, the abundance of earthworms was 32.0–224.0 m^{-2}, 1–5 species, and in the Saaremaa area, the abundance of earthworms was 0–614.0 m^{-2}, 0–5 species. There were differences between Palamuse and Saaremaa pilot areas; however, these differences were statistically insignificant (Table 2).

(4) Although the abundance and number of species was higher in Saaremaa, no differences were found in the earthworm biomass and in the ecological composition of communities between the two areas.
(5) The hydrolytical activity of the microbial community was significantly higher ($p < 0.05$) in Saaremaa than in Palamuse (0.87 ± 0.09 and 0.54 ± 0.05 OD (Optical Density, according to Schnürer and Rosswall 1982) per gram dry soil, respectively). The differences between AEP pilot areas and their reference areas (Lümanda-Kihelkonna: 0.87 ± 0.09; Mustjala: 0.74 ± 0.05) were statistically insignificant.
(6) All calculated soil indicators were above EMR values, which indicates the stability of soil ecosystems (Table V in **IV**).

Table 2. Characteristics of earthworm and microbial communities: the mean value ± SE (**IV**).

Parameter	Palamuse (intensive agriculture)	Saare (reference are the intensive agriculture)	Kihelkonna/ Lümanda (extensive agriculture)	Mustjala (reference area to intensive agriculture
N, individuals per m²	117.1 ± 18.6	111.4 ± 24.2	149.3 ± 58.6	150.7 ± 55.6
S, number of species per m²	3.1 ± 1.0	3.33 ± 1.0	3.3 ± 1.3	3.0 ± 1.5
M, g per m²	45.5 ± 11.4	27.0 ± 8.4	45.4 ± 21.2	82.9 ± 54.2
Body mass of individual, g	0.37 ± 0.03	0.27 ± 0.07	0.39 ± 0.04	0.66 ± 0.03
Dominancy, %	88.2 ± 4.1	90.3 ± 3.5	80.4 ± 5.0	82.9 ± 13.3
Epigaeic earthworms, %	1.0 ± 0.7	0	4.3 ± 2.0	0.8 ± 1.4
Endogaeic earthworms, %	83.2 ± 9.5	95.4 ± 2.3	69.0 ± 11.9	87.7 ± 28.0
Anecique earthworms, %	5.8 ± 2.0	6.7 ± 2.6	6.7 ± 2.6	11.5 ± 8.2
Hydrolytical activity, OD per g dry soil	0.54 ± 0.05	0.57 ± 0.11	0.87 ± 0.09	0.74 ± 0.05

Discussion

The pilot project provided valuable experience in testing AE measures and soil biota indicators in Estonian agricultural landscapes. The soil biota indicators have considerable potential as early indicators of soil degradation or improvement and could be applied for monitoring the effects of AE measures at the field level. However, in this current study, the study time was too short (one year) to detect statistically significant differences between the AEP pilot areas and their reference areas in the same regions.

There were no differences in the abundance and number of earthworm species between AEP pilot areas and their reference areas. The study found minor differences in the abundance and number of earthworm species between intensively managed (Palamuse) and extensive managed (Saaremaa) pilot areas; however, these differences were statistically insignificant. Moreover, considering the difference in their natural conditions and particularly because of different soils, it could be difficult to separate the impacts of farming intensity and contrasting natural conditions on earthworm communities.

The study found statistically significant ($p < 0.05$) differences in the hydrolytical activity of the microbial community between Palamuse and Saaremaa pilot areas; however, no differences were found when comparing the pilot areas with their reference areas in the same county. It was predictable that one year is definitely too short period to evaluate the impact of AEP measures on soil biota. When evaluating the effects of specific measures on soil biota, it is important to consider the time factor, i.e., how long the measures have been applied for. Therefore, the detected differences between Palamuse and Saaremaa pilot areas were most likely determined not only by different farming intensities but also by different natural conditions of the pilot areas.

The study indicates that a prolonged application of AE measures may increase the abundance and diversity of earthworms, decrease the dominance of *A. caliginosa*, and increase the activity of the microbial community (**IV**). Further studies by Ivask *et al.* (2007) have proven that the specific composition of an earthworm community indicates the intensity of agricultural activity at the field level. However, because the effectiveness of AE measures on biodiversity is difficult to assess and because the existing studies are often controversial, more research is required to assess and improve their actual influence on different species. The collected data could be a baseline for future evaluations of the effects of different AE measures.

Weather aspects were also considered because some authors (Fründ *et al*. 2011) have emphasised the dependence of soil biota measurements, including earthworm parameters, on weather conditions. The weather conditions in 2001-2002 were suitable for earthworm communities. The summer (June-September) mean air temperatures were close to normal for these regions, and the soil moisture conditions were optimal for earthworms.

The pilot study demonstrated that soil biota measurements might be sensitive and provide essential feedback if these measurements are integrated in long-term monitoring programs of agricultural landscapes. Regarding the complex nature of landscapes, the data for landscape analysis could be obtained not only from special landscape monitoring programs but also from other environmental monitoring programs. For that reason, the next study (**V**) analyses the spatial distribution of different monitoring networks in Estonian to combine and integrate data, as well as to optimise or to improve the developed monitoring concept for agricultural landscapes.

3.4. Neighbourhood-defined approaches for integrating and designing landscape monitoring in Estonia (V)

The scope of landscape monitoring may vary from surveys of visible land use or land cover patterns that are suitable to reflect landscape change to comprehensive surveys of landscape dynamics that cover a variety of phenomena and interactions (Bastian *et al.* 2002). The Estonian landscape monitoring program has three sub-programs: agricultural landscapes, coastal landscapes and land cover (Table I in **V**), which each have different objectives, and the concept of "landscape" used in monitoring also varies widely. In land cover monitoring, landscape is meant simply as a spatial landscape pattern. However, in terms of the general landscape monitoring concept, landscape is considered a dynamic and interrelated geo-complex of biotic and abiotic components, in which a change in one component affects the entire complex (Arold 1991, 2005). Consequently, landscape monitoring may by focused on its different components, particularly on those components which, according to Bastian *et al.* (2002), have an indicator value that is sufficient to reflect the conditions of the entire landscape. Additionally, in theory, the data for landscape analysis in Estonia could be derived not only from special landscape monitoring programs but also from other environmental monitoring programs, such as biodiversity, forest, soil, water, air and integrated monitoring, which are altogether from 11 monitoring sub-programs (Table II in **V**). The study analysed Estonia's spatial coverage by monitoring data to combine and integrate data from different monitoring sets. The study design and methodology, as well as the basic results, have been published by Roose (2005) and in paper **V**.

Aims of the study

The main objectives of the study were as follows:

- To analyse the distribution of monitoring data by Estonian landscape regions (Arold 2005) and CORINE land cover (CLC) classes.
- To analyse the spatial distribution of Estonian landscape monitoring (3 sets) and other complementary environmental monitoring sets.
- To analyse the possibilities to compile and integrate data from the landscape monitoring and other complementary environmental monitoring sets.

Materials and methods

The data for the spatial analysis of monitoring networks were obtained from landscape monitoring (three sets) and other complementary environmental monitoring sets (11 monitoring themes), which incorporates 1,316 monitoring stations and reports 227 parameters (Table I and II in **V**). In addition, the digital maps of Estonian landscape regions (Arold 2005), soils and CLC classes (Meiner 1999), which were all at an original scale of 1:100,000, were used.

The methodology is based on the statistical description of point patterns and neighbourhood analysis, which were characterised by the nearest neighbourhood index and Ripley's *K*-function (Ripley 1981, Upton and Fingleton 1985, 1989, Haining 2003). In this approach, the main tools are distance parameters or distance statistics. The densities of sets are weighted according to their distance zones and neighbourhoods. Monitoring sets are described by distance methods, which assess distances between points or the distance to geographical objects or factors. The pattern of monitoring is modelled from landscape regions, land cover, or soil classes. MS Excel, MapInfo, Vertical Mapper, and CrimeStat programs (Levine 2002) were used for analyses of the spatial model and data. The thematic maps were produced using the MapInfo Professional software.

The nearest neighbour index is a measure of first-order spatial randomness. This index compares the distances between the nearest points with the distances that would be expected from chance (Ripley 1981). The distance to the nearest neighbour is calculated and averaged over all points.

$$d\left(NN\right)\frac{\sum_{i=1}^{N}\left(\left(\min\left(\mathrm{d}_{ij}\right)\right)/N\right)}{\left(\frac{1}{2}\right)\left(\sqrt{A/N}\right)} \tag{1}$$

where N is the number of points in the distribution, $\min(d_{ij})$ is the distance between each point and its nearest neighbour, and A is the area of the region. If the index is equal to 1.0, then the index indicates random distribution. If the index is greater than 1.0, then there is an evidence of dispersion, and if the index is lower than 1.0, then the index indicates a clustering of the distribution.

Ripley's *K*-function is an upper order nearest neighbourhood statistic, which provides a test of randomness for every distance from the smallest to the size of the study area. Ripley's *K*-function is designed to measure second-order trends (Ripley 1981) or the hierarchy of clustering. As a second-order statistic, Ripley's *K*-function shows how local clustering is compared with a general pattern of the set over the region (O'Sullivan and Unwin 2003). Similar to the nearest neighbour index, Ripley's *K*-function was applied to compare the monitoring sets. Under unconstrained conditions, K is defined as:

$$K\left(d_s\right)=\frac{A}{N^2}\sum_{i}\sum_{j}I\left(d_{ij}\right) \tag{2}$$

$$\mathrm{d}_s=\frac{R}{100} \tag{3}$$

where $I(d_{ij})$ is the number of other points, j, found within the distance d_s, added together over all points, i. R is the radius of a circle for the study area. $K(d_s)$ is transformed into a square root function. $L(d_s)$ is defined as:

$$L\left(d_s\right)=\sqrt{\frac{K\left(d_s\right)}{\pi}}-\mathrm{d}_s \tag{4}$$

A density analysis of monitoring networks was performed in a 50 km search radius. It was assumed that a data transfer function could be applied for this distance, and also, that 50 km could be taken as the average maximum distance between the monitoring stations.

Results

(1) The study determined the distribution of monitoring stations (altogether and by topical sets) by CLC first mapping classes (Meiner 1999), by Estonian landscape regions (Arold 2005), and, for forest monitoring stations, also by soil types, for all stations from 1:100,000 digital maps.

(2) The density of monitoring stations is higher in urban and port areas (17–40 stations per 100 km^2), where the monitoring strategy focuses on human impact. However, the distribution of all sets, in general, reflects the distribution of land cover: 9% of all monitoring stations in Estonia are in built-up areas, 39% in semi-natural areas, 43% in natural areas (excluding wetlands), 6% in wetlands and 3% in lakes and rivers (Table III in **V**).

(3) The representation of monitoring stations by topical sets shows that the monitoring of plants intensively covers alvars in coastal lowlands. Natural grasslands are proportionally "over-represented" due to the targeted monitoring of rare and endangered species.

(4) The forest monitoring set corresponds more or less to a proportional random selection throughout different forest types: 51 stations are situated in coniferous forest, 27 stations in mixed forest, and 9 in broad-leaved forest.

(5) The distribution of the forest monitoring stations by soil types indicates over-representations of rendsic lepsol and skeletic regosol soil types, as well as underrepresentations of stagnic luvisols and dystric histosols.

(6) Due to the topic-based establishment of monitoring networks, the stations are not randomly distributed. For instance, the environmental monitoring stations are concentrated in the Tallinn area and in northeast Estonia, which has a higher human impact, and to a lesser degree in Pärnu and western Saaremaa, which, in turn, are covered by a dense biodiversity monitoring network.

(7) The monitoring network for agricultural landscapes covers agricultural areas in most of the larger landscape regions in Estonian (Fig. II in **V**).

(8) Large landscape regions are proportionally less covered by the total monitoring network, and small regions, such as the coastal lowland of the Gulf of Finland, Karula upland, and Palumaa, are more intensively surveyed. According to the geographical distribution, coastal lowlands have the most intensive coverage by monitoring sets (Fig. II in **V**).

(9) The neighbourhood analysis (characterised by the nearest neighbourhood index and Ripley's *K*-function) indicates regular, dispersed, aggregated, and random patterns in Estonian monitoring sets.

(10) The spatial distribution of nine topical monitoring networks was analysed by the nearest neighbourhood index (Fig. III in **V**). In general, pollution-related sets tend to cluster around 'hot spots', with few reference areas represented. The groundwater monitoring set is the most closely clustered, with a nearest neighbourhood index of 0.18. The set forms clusters in north-eastern Estonia, in Pandivere, which is a nitrate-sensitive area, and in the Tallinn area, with a significant human impact.

(11) The monitoring sets of plant and animal species (flora and fauna) are clustered in protected areas (Fig. III in **V**).

(12) Compared with other sets, the meteorological monitoring set is the most dispersed, and landscape monitoring also shows higher dispersion (Fig. III in **V**).

(13) The forest monitoring set is the most regular. This set is located across Europe on a grid of 16 km×16 km. Estonia has 90 monitoring stations, with 2,136 observation trees. (Fig. III in **V**).

(14) The Ripley's K-function enables the description of the hierarchy of clustering compared with various baseline landscape characteristics (Fig. IV in **V**). Clustering is expressed clearly in the groundwater monitoring set, which has a radius of 30 km for groundwater bodies. Hierarchical clusters are clearly described in the plant species monitoring set, where the density of the point pattern increases to a search radius of 25 km, which represents the size of larger protected areas. The 80 km buffer expresses the distance between nature protection areas. For smaller sets, such as those sets for meteorology and soils, the curve shows an increase in clustering over long distances.

(15) The density analysis of monitoring networks, at a 50 km search radius, determined regional differences in spatial coverage by monitoring data (Fig. 3). Stratified environmental information is provided for landscapes near the Tallinn area, Pärnu and in north-eastern Estonia (Kurtna Lakes). In addition, the Endla Nature Reserve and Viidumäe National Park are certainly covered (Fig. 3). Areas that are more sparsely and less certainly covered by monitoring information are the western-central part of Estonia and the border areas with Latvia. Smaller "uncovered" areas are found in northern Kõrvemaa, in Avinurme, around Varbla and on Peipsi lowland.

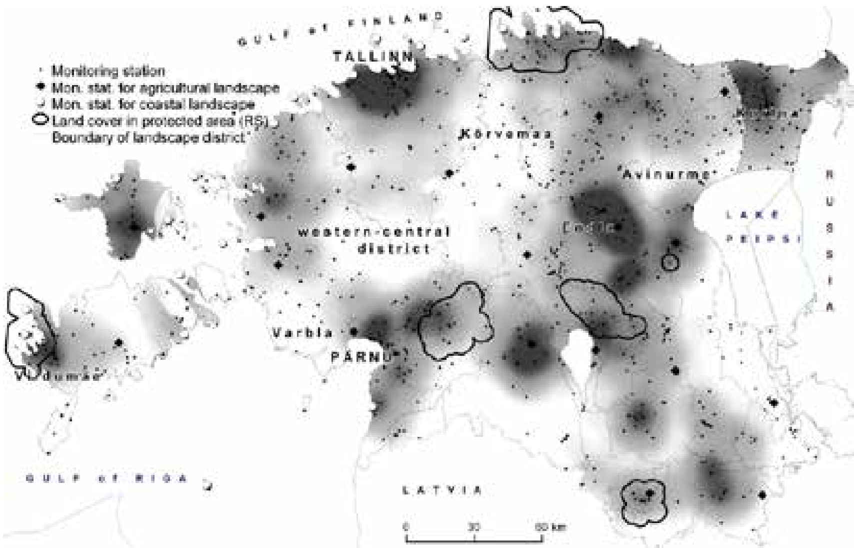

Figure 3. Total density of monitoring sets at 50 km search radius, dark: high density, light: low density (V).

Discussion

The main advantage of the stratified assessment of monitoring data coverage is that, even if the geographic data or some characteristics that the user is looking for are not available, knowledge from other areas and the identified patterns enable a compromise to be made between requirements and availability. The exploration of areas having the same land cover, soil types, etc., enhances data mining techniques from available sampling sets that best suit our objectives.

In Estonia, the topic-based environmental monitoring sub-programs are independent of each other, and the monitoring networks have not been established randomly. A regular monitoring grid is only available for forest monitoring. All other sets are based on their own monitoring objectives, some of which aim to achieve overall national geographical coverage, whereas others aim to test different landscape components or regions. The monitoring sets that aim to acquire data on human impacts are clustered in industrial, urban and environmentally sensitive areas. Water monitoring sets are clustered around river basin areas. Biodiversity sets can easily be applied as data sources for landscape monitoring in national parks and other protected areas (**V**, Roose and Sepp 2010). The representation on landscapes and land cover types is rather different, and the entire complex of stratified monitoring data could not be equally available everywhere in Estonia. Small sets having less than 50 stations are biased, and tests have not found their data to be statistically significant. For that reason, the application of the data transfer functions requires further investigation and modelling on a small and meso-scale level (**V**, Roose and Sepp 2010).

Designing monitoring networks to be spatially more efficient is one of the key aims for upgrading monitoring methods and decision-support systems. A systematic approach that focuses on landscape regions may help to optimise the monitoring sets and, in this way, may achieve a more coherent and efficient layout of monitoring sets across the country. The monitoring sites for agricultural landscapes are chosen to represent the major landscape differences in Estonia. At the time of the study (**V**), there were 18 strategically chosen monitoring sites in Estonia with areas of approximately 4 km^2 each, and, by now, the number has increased to 24 sites, with a total area of approximately 100 km^2. However, concerning network design, there remains a conceptual choice between the chosen strategic approach, which is proposed to be more efficient, and the random network design, which, in turn, gives landscape aspects the opportunity to be chosen by chance and, in this way, helps to exclude subjectivity (Bunce *et al.* 1996, Brandt *et al.* 2002, Bailey and Herzog 2004). For statistical analyses, the random network design has several advantages and does not require that the total monitoring area be increased if the network includes more randomly selected sites of lesser size.

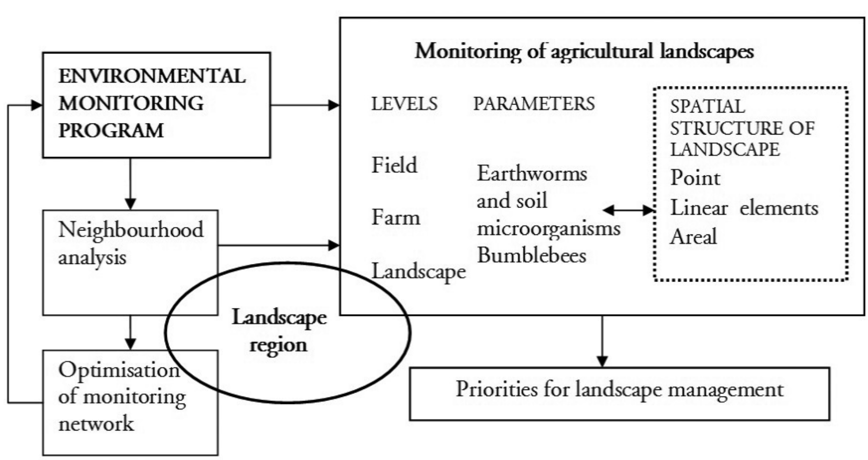

Figure 4. Framework for applied integrated landscape monitoring (V).

The analysis of the spatial coverage of available monitoring data indicates the possibilities to improve the developed monitoring network, as well as the monitoring strategy for agricultural landscapes. In Estonia, the chosen integrated multi-scale landscape monitoring concept for agricultural landscapes already incorporates the monitoring of spatial land use characteristics and biodiversity (soil biota and bumblebees) parameters; however, it could be substantially supported by datasets of environmental monitoring (Fig. 4). Complementary data on landscape components could be obtained from other environmental monitoring sub-programs either directly or by applying different methods of extrapolation, such as the neighbourhood method, which uses the spatial unit of landscape regions.

4. CONCLUSIONS

The thesis findings draw the following main conclusions:

Land use changes in Hiiumaa since the 1950s (**I, III**)

1. The detailed spatial land use analysis that was performed in two main agricultural regions of Hiiumaa and that was based on available large-scale decoded aerial photos from 1956 to 1998 and field studies in 2004/5 allowed the exploration of the transformation of the former mosaic land use patterns that are characteristic of extensive agriculture to more uniform patterns of intensive agriculture. This landscape change is directly linked to the decline in identity and diversity values of traditional agricultural landscapes in Hiiumaa.

2. In Hiiumaa, the greatest change in landscape structure (characterised by landscape metrics) took place by the 1980s, when the traditional and extremely diverse land use pattern had simplified and polarised as a result of the intensification of farming, marginalisation, collectivisation and, in particular, land reclamation, which most likely had the greatest effect on this process. The remarkable changes in agricultural land use intensity since the beginning of the 1990s, which included a sharp decline by the end of the 1990s and a remarkable revival since 2004, have not yet caused any significant changes in developed landscape patterns.

3. The results of the extensive spatial study demonstrate that the chosen methodology allows the detection of changes in the compositional heterogeneity (the number and proportions of different land use types) and in the configurational heterogeneity (the spatial arrangement of different land use types) of rural landscapes, to elucidate the origin of the larger ameliorated field systems and to track the transformation of the once widespread open and half-open semi-natural grasslands to forests and other land use types.

4. The simplification of land use/cover patterns has controversial impacts to biota: e.g., the disappearance of diverse land use patterns and the sharp decrease in half-open farmlands (including traditional wooded meadows and pastures) can be related to biodiversity loss; however, the huge agricultural areas (cultivated grasslands and cereal

fields) have become important feeding habitats for migrating birds (geese and cranes). Due to the long intervals of the spatial analysis (1956–1984–2004–2010), it was difficult to find solid correlations between available annual bird count numbers and changes in land use patterns. The developed homogeneous and spacious land use pattern seems to favour the general state and abundance of migrating (and breeding) crane assemblies; however, the actual land use of the large agricultural areas, i.e., the area under cereals fields and regular mowing of grasslands, tends to have an even greater effect.

5. The developed landscape pattern tends to be more inert compared with the actual annual use of the fields. The renaturalisation of abandoned fields depends not only on the time factor but also on several other factors, such as the size of the fields, the characteristics of soils and neighbouring areas, etc. Therefore, in Hiiumaa, the developed pattern of spacious field systems on thin and poor soils remained quite stable, even after long-term abandonment in the 1990s.

The impact of land use changes on autumn staging cranes (**II**)

6. The following study demonstrates that the local differences in autumn staging crane numbers predominantly depend on changes in agricultural land use in staging areas, rather than on more stable landscape patterns. The annual differences in local count numbers in Hiiumaa and Matsalu most likely indicate the relocation of birds to other staging areas as determined by available food sources.

7. The study determined a significant positive correlation between the number of staging cranes and the cropping area of winter rye, winter wheat, summer wheat and all cereals combined, and a negative correlation between crane numbers and the cropping area of potato was also found.

Soil bioindication of AE measures (**IV**)

8. The short-term application of AE measures cannot have significant effects on soil biota. The pilot project provided valuable experience in testing AE measures and soil biota indicators in Estonian agricultural landscapes; however, in this current case, the study time (one year) was definitely too short to detect statistically significant differences between the AEP pilot areas and their reference areas in the same regions.

9. The AE measures can have positive effects on soil biota and certain soil bioindicators (the abundance, diversity and ecological composition of earthworm communities, as well as the hydrolytical activity of the microbial community) can be used for evaluating the effectiveness of the AE measures, as well as the level of human pressure, on the field level when integrated in long-term monitoring programs of agricultural landscapes. The study found statistically significant ($p < 0.05$) differences in the hydrolytical activity of the microbial community and minor differences in earthworm parameters between Palamuse (intensive) and Saaremaa (extensive) pilot areas. However, in this study, it was impossible to separate the impacts of farming intensity and contrasting natural conditions on soil biota, and the detected differences (between Palamuse and Saaremaa pilot areas) were most likely determined by both of these factors.

Analysis of Estonian monitoring networks (**V**)

10. The neighbourhood analysis can be used for analysing the spatial coverage of developed monitoring networks, designing monitoring networks to be spatially more efficient and upgrading monitoring methods and decision-support systems. A systematic approach that is focused on landscape regions may help to optimise the monitoring sets, and, in this way, to achieve a more coherent and efficient layout of monitoring sets across the country.

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SUMMARY IN ESTONIAN

Maakasutusmuutuste mõju maapiirkondade maastikulisele ja bioloogilisele mitmekesisusele

Viimase aja teaduskäsitlustes vaadeldakse maastikku ajas kulgeva protsessina, mis kujuneb looduslike ja sotsiaalmajanduslike tegurite mõjul (Palang 1998), ning muutumist maastiku olulise omadusena (Antrop 2003, Carranza jt 2007). Seepärast ei ole põhjust eristada termineid „maastik" ja „maastikumuutus" ning maastiku- ja maakasutusmuutused on saanud tähtsaks uurimisteemaks kõikjal maailmas.

Eestis on uuritud maastikumuutuste erinevaid tahke, näiteks Varep (1964) ja Arold (1991, 2005) on analüüsinud Eesti maastiku regionaalsete erinevuste kujunemist, Hellström (2002) on uurinud üldiseid tendentse Hiiumaa põllumajandusmaastike ja hoonestuse kujunemisel, Mander ja Palang (1994, 1999) on toonud välja 20. sajandi maastikumuutuste üldised suundumused ning juhtivad mõjurid kogu Eestis. On tehtud mitmeid uuringud, mis käsitlevad näiteks muutusi maastiku tunnetuslikus väärtuses (nt Palang jt 2011), püüavad prognoosida tulevikus toimuvaid muutusi (Palang 1998; Palang jt 2000, 2010), kasutavad kaugseire meetodeid muutuste tuvastamiseks Eesti metsades (Püssa jt 2005, Liira jt 2006, Peterson jt 2006) või põllumajanduslikus maakasutuses (Peterson ja Aunap 1998). Hoolimata kirjanduse suurest hulgast on vähe uuringuid, mis oleksid suunatud põllumajandusmaastike muutuste ja elustiku vahelistele seostele, põhineksid suuremõõtkavalistel kaardimaterjalidel ning keskenduksid viimase poolsajandi poliitiliste ja sotsiaalmajanduslike reformide mõjule. Peale ühekordsete uuringute on tekkinud ka vajadus juba alanud riikliku põllumajandusmaastiku seireprogrammi (Sepp ja Kaasik 1995) edasiarendamiseks.

Käesolev doktoritöö analüüsib artiklite **I–V** põhjal viimasel poolsajandil Hiiumaa põllumajandusmaastikus toimunud muutusi (**I**), keskendudes eeskätt muutustele avatud ja poolavatud maakattetüüpides (**III**). Käsitlemist leiavad ka põllumajandusliku maakasutuse ja elustiku vahelised seosed, s.o viimaste aastakümnete maakasutusmuutuste võimalik mõju sügisesel läbirändel Eestis peatuvate sookurgede (*Grus grus*) arvukusele (**II**) ja mulla bioindikaatorite sobivus põllumajandus- ja keskkonnameetmete tõhususe ning inimmõju taseme hindamiseks (**IV**). Põllumajandusmaastiku seirekontseptsiooni ja -võrgustiku arendamiseks ning täiendavate seireandmete lõimimiseks analüüsiti Eesti teemapõhiste seireprogrammide andmestike territoriaalset jaotumust erinevate maastikutunnuste järgi (**V**).

Hiiumaa maakasutusmuutuste uuring (**I** ja **III**) hõlmas saare kahte peamist põllumajanduspiirkonda: Hellamaa (200 km^2) saare kirdeosas ja Vanamõisa (67 km^2) lõunaosas. Maastikuanalüüs põhines 1956. ja 1984. aasta suuremõõtkavalistel (1 : 10 000) ortofotoplaanidel, Eesti digitaalsel põhikaardil ja ulatuslikel välitöödel 2004.–2005. aastal. Maastikumustrite ülepinnalisel digitaliseerimisel kasutati põllumajanduslike majandite maakasutuplaanide klassifikatsiooni, mis eristab enam kui 30 maakasutustüüpi. Suksessiooniliste üleminekute uurimiseks märgiti rohumaade, karjamaade ja soode puhul ka nii puude kui ka põõsaste esinemine. Detailne maastikuanalüüs võimaldas uurida traditsioonilise, s.o ekstensiivsele põllumajandusele iseloomuliku mitmekesise ja mosaiikse maakasutusmustri olulist lihtsustumist ning polariseerumist Hiiumaal. Suurim muutus maastikumustris leidis aset juba 1980. aastateks, peamiselt uute intensiivseks põllumajanduseks sobivate maade raadamise ja sobimatute marginaliseerimise tõttu. Ajavahemikul 1956–1984 vähenes põllumajandusmaa valdavalt looduslikku tüüpi rohumaade arvel kokku umbes 43%, samal ajal suurenes metsade pindala umbes 44%. Järgnevad suured muutused põllumajandusmaa kasutuses: järsk vähenemine 1990. ja elavnemine 2000. aastatel ei ole seevastu toonud kaasa suuri muutusi juba väljakujunenud lihtsustunud maastikumustris.

Uuring aitas selgitada praeguste põllumassiivide päritolu ning kunagi laialt levinud heina- ja karjamaadega toimunut. Näiteks 1956. aasta heina- ja karjamaadest oli 1998. aastaks üle poole kaetud metsaga, kusjuures ainult 3% karjamaadest ja 12% heinamaadest olid säilitanud oma varasema ülesande. Valitud metoodika võimaldas kaudselt hinnata ka muutusi liigirikaste puisniitude ja -karjamaade levimuses. Kui 1956. aastal olid poolavatud põllumajandusmaad, s.o nii puude kui ka põõsastega hajusalt kaetud heina- ja karjamaad levinud kokku 7492 hektaril, siis 1984. aastaks oli taoliste maade pindala kahanenud enam kui kümme korda 695 hektarile. 1998. aastaks oli poolavatud alade pindala suurenenud 1072 hektarile, kuid seda peamiselt hüljatud põllumajandusmaade renaturaliseerimise arvelt. 2010. aasta andmetel leiti ELi loodusdirektiivile vastavaid Fennoskandia puisniite ja -karjamaid (6530 ja 9070) vaid Hellamaa uurimisalal – kokku vastavalt 42 ja 112 hektaril (ENIS 2010).

Maastikumuutuste mõju elustikule on vastuoluline. Traditsioonilise maakasutusmustri kadumine ja poolavatud põllumajandusmaade oluline vähenemine on kahandanud liigirikaste elupaigatüüpide levimust, samas on suured põllumassiivid saanud tähtsaks toitumis- ja peatuspaigaks paljudele läbirändavatele linnuliikidele. Hiiumaal läbi viidud mastaapse maastikuanalüüsi üks eesmärke oli uurida ka võimalikke korrelatsioone muutunud maastikumustrite ja rändlindude iga-aastaste loendusandmete vahel. Maastikuanalüüsi pikkade ajaliste intervallide (1956–1984–1998–2004) ja maakasutusmustrite suurema püsivuse tõttu ei õnnestunud otseseid seoseid siiski leida, küll aga leiti järgneva uuringuga statistiliselt olulisi korrelatsioone teatavate põllukultuuride külvipindade ja sügisel Eestis peatuvate sookurgede (*Grus grus*) loendusandmete vahel (**II**), kusjuures Hiiumaal oli nimetatud seos kõige tugevam talirukki ($r_s = 0{,}58$, $p < 0{,}05$, $n = 21$) ja talinisu ($r_s = 0{,}58$, $p < 0{,}05$, $n = 21$) külvipindadega ning Matsalus kõigi teraviljade ($r_s = 0{,}56$, $p < 0{,}05$, $n = 28$) koondkülvipinnaga.

Maakasutusmuutuste ja elustiku vahelisi seoseid uuriti ka bioindikatsiooni aspektist. Katseprojekt, mis viidi läbi põllumajandus- ja -keskkonnameetmete mõju uurimiseks mullaelustikule, püüdis samas selgitada ka mulla teatud bioindikaatorite sobivust meetmete tõhususe hindamiseks (**IV**). Uuring viidi läbi 2001.–2002. aastal kahel katsealal: Palamusel, mis esindas intensiivset põllumajandust, ja Kihelkonna-Lümandus, mis esindas ekstensiivset põllumajandust, ning nendega samas piirkonnas olevatel võrdlusaladel. Uuring ei tuvastanud statistiliselt olulisi erinevusi katsealade ja võrdlusalade vahel ning näitas ilmekalt, et lühiajaline põllumajandus- ja keskkonnameetmete rakendamine ei avalda mullaelustikule suurt mõju. Uuring näitas, et põllu tasemel võivad mulla teatud bioindikaatorid (vihmaussikoosluste parameetrid ja mullamikroobide hüdrolüütiline aktiivsus) anda tähtsat teavet mulla seisundi ja inimmõju kohta, kui need on lõimitud pikaajalisse põllumajandusmaastike seiresse. Juhtuuring tuvastas statistiliselt olulise ($p < 0{,}05$) erinevuse mullamikroobide hüdrolüütilises aktiivsuses intensiivselt ja ekstensiivselt majandatud katsealade vahel. Arvestades katsealade kontrastseid loodustingimusi Palamusel Jõgevamaal ja Kihelkonna-Lümandus Saaremaal, tulenes nimetatud erinevus mitte ainult majandamise erinevast intensiivsusest, vaid suurel määral ka erinevate loodustingimuste mõjust, kuid selle uuringu raames ei olnud võimalik nende kahe peamise teguri mõju eristada.

Maastikuökoloogias mõistetakse maastikku dünaamilise ja hierarhilise geokompleksina, mille elus ja elutud komponendid on vastastikku seotud nii oma arengus kui ka ruumilises paiknemises (Arold 2005). Sellisest käsitlusest lähtuvalt ei pruugi maastikuseire piirduda üksnes maakatte- või maakasutusmustrite analüüsiga, vaid lisateavet maastikus toimuva kohta võiks saada ka autonoomselt arendatud maastikukomponentide seirest. Põllumajandusmaastiku seirekontseptsiooni arendamiseks ja sellesse täiendavate seireandmete lõimimiseks analüüsiti Eesti teemapõhiste seireprogrammide andmestiku territoriaalset jaotumist erinevate maastikutunnuste, näiteks Eesti maastikurajoonide, CORINE maakattetüüpide ja mullatüüpide järgi (**V**). Seirevõrgustike ruumilist paiknemist iseloomustati ka naabrusanalüüsi, s.o lähimnaabruse indeksi ja Ripley K-funktsiooni ning tihedusnäitajate abil. Analüüs hõlmas kokku 14 erinevat maastiku-, elustiku- ja keskkonnaseire alamprogrammi. Teemapõhiste seirevõrgustike puhul ei saa eeldada ühtlast juhuslikku paiknemist, sest nende arendamisel on lähtutud oma sisemisest loogikast ja otstarbekusest, nt keskkonnaseire võrgustik on koondunud Tallinna ümbrusse ja Kirde-Eestisse, kus inimmõju on tugevam, ning vähem Lääne-Eestisse, kus omakorda on tihedam elustikuseire võrgustik. Suurim klasterdumine on iseloomulik põhjavee seirevõrgustikule, metsaseire võrgustik on kõige korrapärasem ja meteoroloogilise seire võrgustik kõige hajusam. Seirevõrgustike tihedusanalüüs 50 km otsinguraadiuses näitas, et mitmekülgne seireteave ei ole kõikjal Eestis ühtviisi kättesaadav: paremini on kaetud näiteks Tallinna ja Pärnu ümbrus, Kurtna maastikukaitseala, Endla ja Viidumäe looduskaitsealad ning Vilsandi rahvuspark; vähem aga näiteks Kesk-Eesti, Põhja-Kõrvemaa, Avinurme ja Varbla kant ning piiriäärsed alad. Põllumajandusmaastike seiresse on juba lõimitud teatavate bioindikaatorite (kimalaste, vihmausside, mullamikroobide) seire, kuid seirekontseptsiooni saaks oluliselt täiendada veel teistegi keskkonnaseire andmestikega. Seireandmete ekstrapoleerimiseks võib kasutada andmete jaotumist teatud ühetaoliste maastikutunnuste, näiteks nii maastikurajoonide kui ka mullatüüpide järgi. Põllumajandusmaastiku seirevõrgustiku kujundamisel tuleb teha kontseptuaalne valik kahe põhimõtteliselt erineva lähenemise vahel. Ühelt poolt seirealade teadlik strateegiline valik, mida peetakse efektiivsemaks, ja teisalt juhuslik seirealade valik, mis iseenesest ei eelda suuremat seireala pindala, kuid aitab välistada subjektiivsust ning sobib seireandmete statistiliseks analüüsiks ja tulemuste üldistamiseks.

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I

ORIGINAL PUBLICATIONS

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TRANSFORMATION OF RURAL LANDSCAPES IN HIIUMAA SINCE 1956 AND THE CONSEQUENCES TO OPEN AND HALF-OPEN SEMI-NATURAL HABITATS

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TRANSFORMATION OF RURAL LANDSCAPES IN HIIUMAA SINCE 1956 AND THE CONSEQUENCES TO OPEN AND HALF-OPEN SEMI-NATURAL HABITATS

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Abstract

Kaasik A., Sepp, K., Raet, J. Kuusemets, V. Transformation of rural landscape in Hiiumaa since 1956: consequences to open and half-open semi-natural habitats. Ekológia (Bratislava), Vol. 30, No. 2, p. 257–268, 2011.

Landscape is a reflection of natural and socio-economic processes. When socio-economic situations change, this alters land use and leads to changed land use patterns. An extensive spatial study of rural landscapes in Hiiumaa Island in West Estonia, based on available aerial photos from 1956 to the present has highlighted significant changes. These include the simplification and polarisation of land use patterns, an overall decrease in agricultural land, especially that of natural and wooded grasslands, and also a gradual increase in forests. The greatest change in land use pattern took place by the beginning of 1980s, then the initial traditional and diverse landscape pattern had become much more simplified and polarised as a result of collectivisation, land reclamation and the wider use of industrial methods in agriculture. The remarkable changes in land ownership and agricultural land use intensity, since the end of the 1980s, have not yet caused any significant changes in developed landscape patterns. This paper examines transformations in the traditional high value farmlands of semi-natural grasslands and wooded meadows and pastures, and the impacts and possibilities of recent political and socio-economic reforms in the preservation of traditional landscapes and biodiversity in rural areas. These factors involve land re-privatisation since 1987, reclamation of independence in 1991 and accession to the EU in 2004.

Key words: rural landscapes, land use change, semi-natural habitats, Hiiumaa

Introduction

Background

Landscape change in Estonia has been well documented during the past century and it followed general European trends (Vos, Klijn, 2000). Existing studies have elucidated the main tendencies of these changes, including the simplification and polarisation of land use patterns, the considerable increase in forests and decrease in agricultural lands and the continual decline in grasslands. They have also discussed the main socio-economic factors behind these land use changes (Mander, Palang, 1994; Palang et al., 1998). Some of the driving forces such as the land reforms of 1919, 1940, 1947 and 1989, deportations and collectivization in the 1940's and the formation of a military border zone along the coastline are specific for Estonia. However, more general factors including the concentration of agriculture, marginalization, land amelioration and the use of larger machines have reshaped rural landscapes in many other countries and contributed to the loss of valuable semi-natural land-use types such as wooded and coastal meadows (Ihse, 1995; Luoto et al., 2003). Unlike previous similar studies (e.g. Palang et al., 1998; Koppa, 2006; Tomson, 2007, etc.), this current land use analysis focuses on the last half-century, especially on agricultural landscapes, and it is based on large-scale cartographic material (1:10 000) and also on extensive field work.

Why Hiiumaa?

Hiiumaa at approximately 1000 km^2 is the second largest island in Estonia. Due to its relative isolation and poor preconditions for intensive agriculture due to young and mostly stony and thin limestone soils, the landscape changes in Hiiumaa have been slower and the land use had retained a relatively traditional character by the middle of the 1950's. So it was presumed that the first available aerial images from 1956 would reflect the diverse land use patterns that had been developed during the first independence period of Estonia from 1918 to 1939.

Fig. 1. Location of study areas in Hiiumaa.

The study area embraces two agricultural regions in Hiiumaa: Hellamaa at 200 km^2 in the north-eastern and Vanamõisa with about 67 km^2 in the southern part of the island (Fig. 1). In Hiiumaa, the amount of agricultural land has drastically decreased from more than 65% in 1939 to less than 25% in 1992 (Mander, Palang, 1999) and currently most of the agricultural land use in Hiiumaa is concentrated in the above regions.

Methodology

The landscape analysis was based on decoded aerial photos (ortophoto maps) from 1956 and 1984. The large-scale ortophoto maps of 1:10,000 were scanned and the land use patterns of 1956 and 1984 were digitized in GIS software MapInfo. The state of the present land use was identified by the 1:10,000 digital Basic Map of Estonia based on 1998 aerial images. Field work subsequently identified the actual land use, namely the fields still in use and long and short-term fallows, thus distinguishing more than 30 land use types. In order to define the former spread of semi-natural wooded meadows and pastures and also to analyze successive transformations between land use categories, the land use types were digitized as precisely as possible, e.g. concerning grasslands, pastures and mires the existence of bushes and trees was defined. The current distribution of preserved valuable open and half-open semi-natural habitats listed in EU Habitats Directive was detected from data of the Estonian Nature Information System (ENIS, 2010). The land use patterns were analyzed and the first statistics and landscape metrics were calculated in MapInfo.

Findings

Simplification and polarisation of landscape patterns

This study has brought out considerable changes in rural landscapes and the first results have already been published (Kaasik et al., 2008). The greatest change in total patchiness and edge density of the land use patterns occurred between 1956 and 1984. By 1984, the traditional and extremely diverse landscape pattern of 1956 had become much more simplified and polarised as a result of farming intensification, marginalisation, collectivisation and especially land reclamation, which most likely had the greatest effect on this process (Fig. 2). In Hiiumaa the amelioration of agricultural lands commenced in 1963 and continued until the beginning of the 1980's, although the major portion of reclamation works involving more than 90% of all reclaimed areas was implemented between 1968 and 1979. By 1998, the landscape pattern and patchiness had remained almost the same as it was in 1984.

In order to analyze the impact of land reclamation on the landscape structure, the larger ameliorated fields and cultivated grasslands (in total 163 fields with a total area of 5,335 ha) were selected (Fig. 3). The spatial analysis inside the ameliorated areas showed a significant simplification in land use structure. The 163 large fields were transformed from 6,167 tiny land use patches in 1956 in the following manner: (1) grasslands and pastures formed 53% with 2920 patches, (2) former fields composed 30% with 2,127 patches, (3) forests comprised

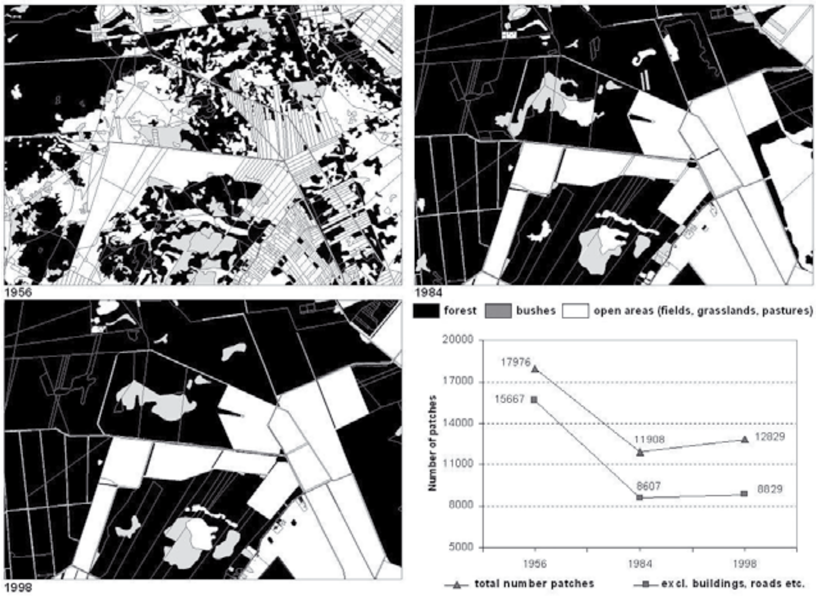

Fig. 2. Changed patchiness and examples of changed land use patterns 1956–1998 (Kaasik et al., 2008).

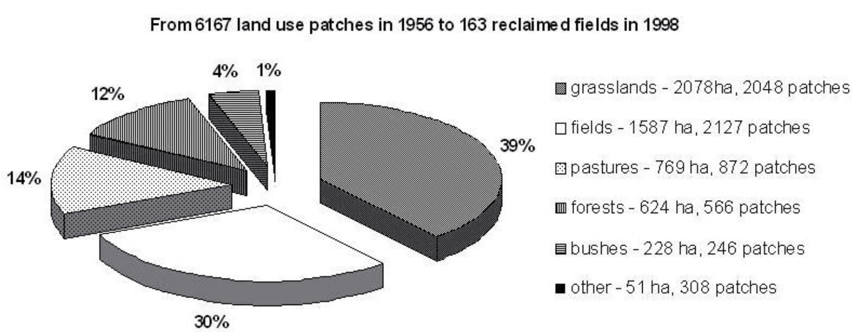

Fig. 3. Origin of 163 ameliorated fields (5,335 ha) in Hiiumaa (Kaasik et al., 2008).

12% with 566 patches and (4) bushes made up 4% (Fig. 3). Other land use types such as roads, courtyards, mires and water bodies together formed 1%. However 9 patches of mires covering 4 ha and 26 patches of water bodies with 7 ha have disappeared due to field land reclamation.

Decrease in agricultural land and increase in forests

An additional tendency was an overall decrease in agricultural land (in account of grasslands) and an increase in forests (Fig. 4a, b). The percentage of grasslands and pastures has decreased significantly, mainly in half-open land use types such as grasslands with bushes and/or trees in our classification (Fig. 5). The sharp decline in half-open farming lands also indicates the loss of species-rich semi-natural land use types and habitats, including wooded meadows and pastures which previously formed as a result of long-term traditional agricultural practices.

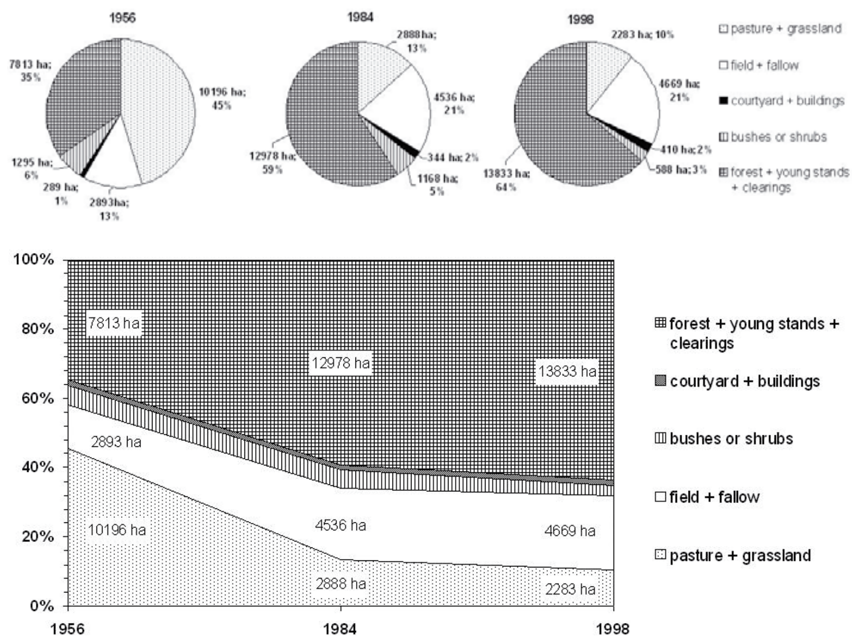

Fig. 4a, b. Changed land use 1956–1998.

Disappearance of wooded meadows and pastures

In 1956, the half-open land use types, such as grasslands and pastures with bushes and/or trees, were widely spread in both study areas, on a total of 7492 ha (Fig. 6). It is now difficult to identify how much of these half-open lands was actually long-term wooded meadows or only recently abandoned farmlands resulting from World War II, deportations and collectivisation. By 1984, only about one tenth of these land use types, comprising 695 ha, were

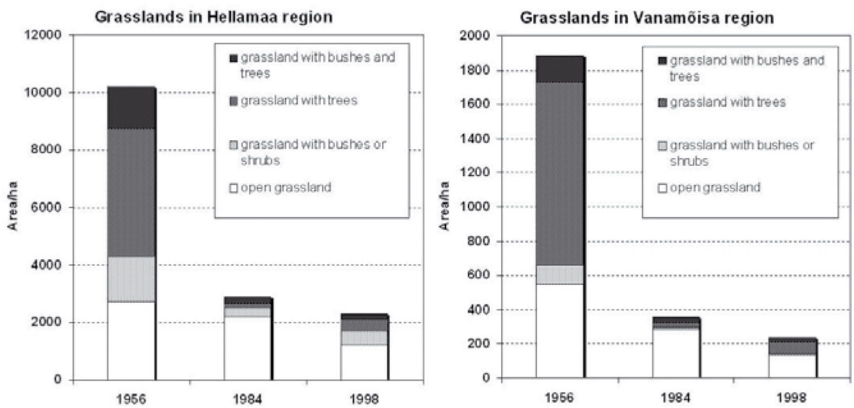

Fig. 5. Decline in grasslands 1956–1998 (Kaasik et al., 2008).

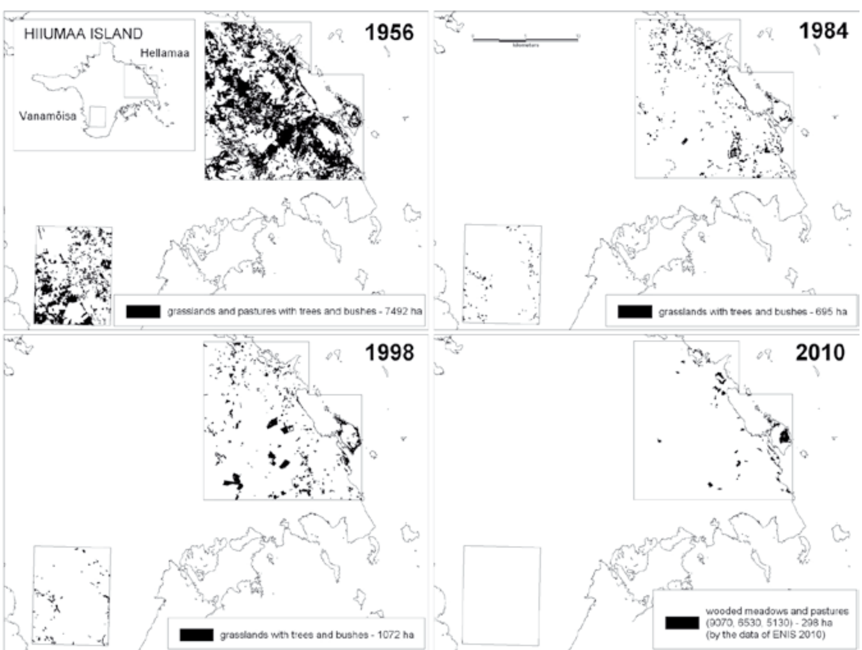

Fig. 6. Disappearance of wooded grasslands in Hiiumaa 1956–2010 (Habitat types according to EU Habitats Directive).

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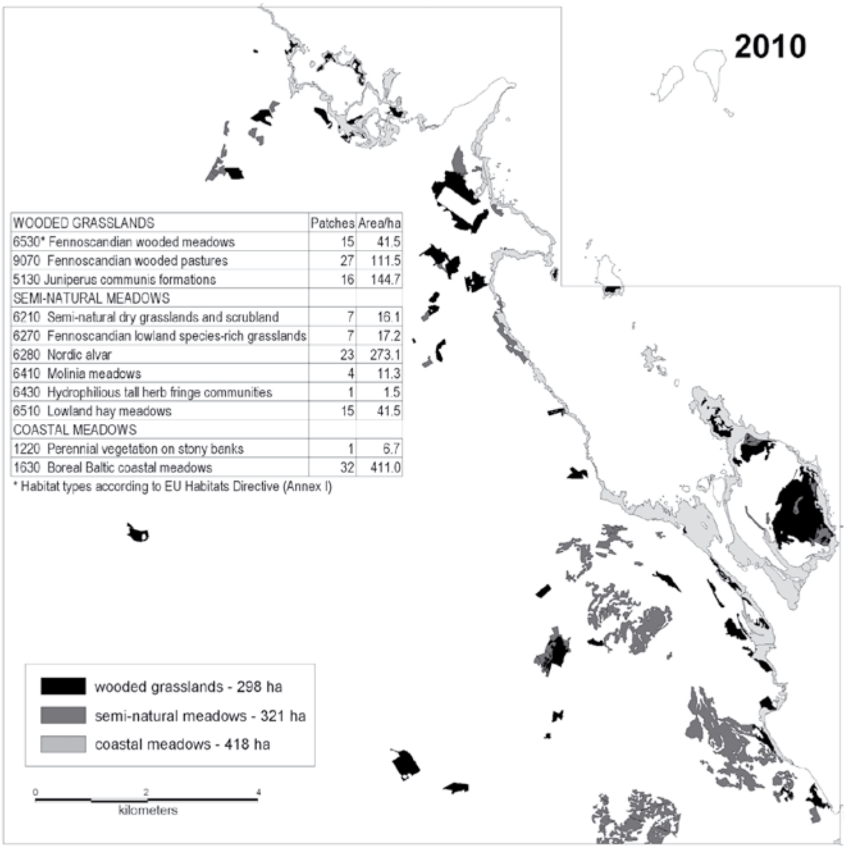

Fig. 7. Current distribution of open and half-open semi-natural habitats in the Hellamaa region (From data of ENIS, 2010; habitat types according to EU Habitats Directive).

preserved mainly due to marginalisation, forestation and amelioration. By 1998, the area of half-open farmlands of 1072 ha slightly increased most likely due to secondary afforestation of abandoned fields and grasslands. By that time, agricultural land use had probably attained its lowest level as a result of reorganization of agriculture involving the collapse of collective farming, land re-privatisation and the formation of small-scale private farms from the end of the 1980's (Peterson, Aunap, 1998). In fact, a great portion of the agricultural areas was fallowed by the end of the 1990's. According to the Estonian Rural Development Plan of 2004–2006, the use of agricultural land in Hiiumaa decreased by up to 50% from 1993 to 2001 (ERDP, 2005).

This general trend leading to the disappearance of extensive half-open land use types has still continued. The present situation only indicates the distribution of valuable half-open habitat types as listed in the Annex I of EU Habitats Directive. These are the Fennoscandian wooded meadows and pastures (6530, 9070) and the *Juniperus communis* formations on heaths or calcareous grasslands (5130) as in Fig. 6. According to data of the Estonian Nature Information System (ENIS, 2010), the valuable open and half-open habitat types have now been preserved only in the coastal region of the Hellamaa study area. These comprise wooded meadows and pastures on 298 ha, semi-natural meadows (6210, 6270, 6280, 6410, 6430, 6510) on 321 ha and coastal meadows (1220, 1630) on 411 ha (Fig. 7).

What has happened to former pastures and grasslands?

Although there are no firm relationships between landscape metrics and ecological values, the shape and size indices/parameters of pastures and grasslands of 1956 indicate their semi-natural character. The mainly used shape metrics are based on perimeter/area ratio (Wu et al., 2000), such as:

$$SI = \frac{0.25P_i}{\sqrt{A_i}}, \tag{1}$$

where *Pi* is the perimeter, *Ai* the area and *SI* the shape index of an individual patch *i*. *SI* estimates the individual shape complexity or compactness. The minimum value is about 0.89 related to the circle and 1.0 for the square. The mean shape index (*MSI*) is calculated for a set of patches composing a class or for a whole area, where *N* is the number of patches within a class or a whole area:

$$MSI = \frac{\sum_{i=1}^{N} \frac{0.25P_i}{\sqrt{A_i}}}{N} \tag{2}$$

The total area of pastures in 1956 was 4,505 ha, with the number of patches (*N*) = 3102, a mean size (*MS*) of 1.45 ha, *SI* – 0.92–4.02 and *MSI* – 1.34. By 1984, 50.4% representing more than half of the pastures were transformed into forests and 13.4% into fields. The transformation of open grasslands, with a total area of 5,706 ha, *N* – 4192, *MS* – 1.36 ha, *SI* – 0.92–5.66 and *MSI* – 1.38 in 1956, was quite similar. By 1984, 43.7% of the grasslands were transformed into forests and 24.4% into fields. From 1984 to 1998, the transformation of previous pastures and grasslands mainly continued by successive transformations, with the gradual overgrowth of bushes and trees. By 1998, more than 56% of the pastures and about 51% of grasslands present in 1956 had been transformed into forests, and only about 3% of previous pastures and 12% of grasslands were maintained and classified as open grasslands.

Discussion

Change is considered an essential property of landscape (Antrop, Van Eetvelde, 2008), and since changing natural or socio-economic conditions alter land use, these are sooner or later reflected in changed land use patterns. Additionally, quantitative changes described by landscape metrics should influence qualitative parameters, such as the historical, aesthetical and biodiversity values of landscapes. However, landscape metrics are difficult to relate to both the processes which cause the change and the qualitative impacts/consequences of the changed land use patterns (Li, Wu, 2004). Due to the complexity of landscapes, the impacts of broader processes can be divided into direct or indirect impacts. These may be positive or negative and some are intended while others are not. This raises the question of how to to direct landscape change. By understanding the general mechanisms which reshape our landscapes and knowing the values and/or features we want to preserve, is it possible to direct these processes?

In Hiiumaa, Soviet period changes brought about the loss of traditional diverse land use patterns that had formed as a result of long-term traditional farming practices. Fortunately, due to unfavourable conditions for intensive agriculture, semi-natural farmlands, including wooded meadows and pastures, are still apparent in many places. When comparison is made with intensive farmlands and economic aspects are, disregarded, the scenic, biodiversity, cultural and potential tourist and recreational values of these landscapes are still considerably higher. There were great expectations that the latest reforms of land re-privatisation and the formation of small-scale private farms from 1987 would help to restore the diverse land use pattern that was characteristic for the first independence period. In fact, a great portion of the privatised agricultural areas remained fallowed for short or longer periods (Raet et al., 2010). As a general trend, until 2004 land use changed towards marginalisation and re-naturalisation of agricultural areas.

Accession to the EU in 2004 opened several subsidiary schemes for Estonian farmers. Some of these were financed by the EU and others by the Estonian Government as part of the National Rural Development Plan. Subsidiary schemes in Estonia are implemented by the Agricultural Registers and Information Board (ARIB). The main objective of a general Area-related Aid and Crop Farming Aid is to preserve open landscapes and to compensate the costs of maintaining land so that its fertility is retained. Several other subsidiaries are designed to improve the environmental awareness of agricultural producers. These are aimed at preserving semi-natural biotic communities and valuable landscape features and maintaining land use. They will also compensate costs incurred in regions with unfavourable or restricted environmental conditions. As preservation of semi-natural habitats and landscape features needs continuous management, and this can not be achieved by traditional nature conservation methods, certain aid schemes are intended to support the reconstruction of stone fences and the management of semi-natural communities. This will supply 238 EUR/ha for wooded meadows and 185 EUR/ha for other ones in 2010. Additionally, there will be encouragement for such undertakings as organic farming and environmentally friendly management (ARIB, 2010). The variety of subsidiaries for Estonian farmers, together with structural changes in the rural economy as more organic and recreational farms join profit-oriented agricultural ones, certainly provides hope that current landscape changes will ensure a more diverse and well-managed rural scene.

Several studies have proven the close relationship between landscape change and biodiversity, wherein a decrease in species-rich wooded meadows and pastures is directly linked to biodiversity loss (e.g. Burel et al., 2004; Grashof-Bokdam, Van Langevelde, 2004; Hietala-Koivu et al., 2004; Bergman et al., 2004). There is no doubt that significantly changed land use and landscape patterns in Hiiumaa have influenced species composition, population numbers and the distribution of several species living, feeding or nesting in these landscapes. The assembled spatial information on landscape change affords further challenges to study the effects of altered agricultural landscapes on biota, and also to calculate potential population numbers of certain species by comparing their habitat preferences with distinct land use types.

As with all changes, the Soviet period landscape change of expansive land reclamation and the formation of huge fields was not completely harmful for all species. There are always so-called winner-species which can adapt, adjust, and even profit from changed conditions. Hiiumaa's agricultural region is an important place for many waterfowl and other species with a most impressive numbers of birds there during spring and in the autumn migration. For instance, cranes and geese on their migration route feed almost exclusively on the cereal fields, cultivated grasslands and pastures of agricultural land. Their breeding success and the species' general state and multitude are considered to be dependent to a large extent on these agricultural resources. The main interest in the large-scale spatial land use data of the last 50 years was induced by the ambition to compare and combine it with available biodiversity data. This especially applied to the existing bird census data from the same period. Due to the long intervals of the spatial analysis of 1956–1984 and 2004–2010, it is difficult to find solid correlations between available annual bird count numbers from the 1960's and changes in land use patterns. Nevertheless, correlation between statistical data for cereal production and the count numbers of the Eurasian crane (*Grus grus*) has already been confirmed (Leito et al., 2008). There is still a lot of uncertainty whether, and to what extent, the changed numbers of some bird species counted in Hiiumaa from the 1960's actually indicated the state of the entire migrating populations or only temporary relocations to new or different feeding and staging areas.

Conclusion

(1) The rural landscapes in Hiiumaa have changed considerably since 1956.

(2) The land use pattern has simplified and polarised as a result of marginalisation, land reclamation, collectivisation and intensification of farming. As the major portion of reclamation works were carried out between 1968 and 1979, the greatest change in landscape patterns took place between 1956 and 1984. In 1998, the general land use pattern remained approximately the same as in 1984.

(3) The area of agricultural lands decreased in grasslands by approximately 43% and forest areas increased by about 44% from 1956 to 1998. Here, the percentage of grasslands decreased significantly especially in natural types of grasslands. By 1998, more than half of the pastures and grasslands of 1956 had been transformed into forests.
(4) The area of half-open land use types such as grasslands and pastures with trees and bushes decreased significantly by more than 10 times from 1956 to 1984. The slight increase from 1984 to 1998 is most likely related to the abandonment of agricultural areas. In 2010, the valuable types of half-open habitats of wooded meadows and pastures have been preserved mainly in the coastal region of the Hellamaa study area which has remained untouched by land reclamation.
(5) The ameliorated fields were mainly transformed to grasslands and pastures (53%), former fields (30%), forests (12%) and bushes (4%). The 163 larger fields existing in 1998 were transformed from more than 6000 land-use patches in 1956.
(6) As a result of land re-privatisation and the formation of small-scale farms since 1987, large portion of farmlands remained fallowed for short or longer periods. The general trend in land use between 1987 and 2004 was abandonment and forestation of farmlands.
(7) Enforcement of the EU agricultural subsidiary schemes since 2004 provides the opportunity to direct current and future landscape change. Aid schemes have been designed to support the reconstruction of stone fences, the management of semi-natural communities and also to assist organic farming and environmentally friendly management.
(8) The decrease in species-rich wooded meadows and pastures is directly linked to biodiversity loss. The accumulated spatial information of landscape change affords further challenges to study the effects of altered agricultural landscapes on biota. Due to the long intervals in spatial analysis between 1956–1984 and 2004–2010, it was difficult to find solid correlations between the annual bird count numbers from the 1960's and landscape metrics. Nevertheless, correlation between statistical data for cereal production and the count numbers of the Eurasian crane (*Grus grus*) has already been confirmed (Leito et al., 2008).

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II

Leito, A., Truu, J., Õunsaar, M., Sepp, K., **Kaasik, A.,**
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THE IMPACT OF AGRICULTURE ON AUTUMN STAGING
EURASIAN CRANES (*GRUS GRUS*) IN ESTONIA

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The impact of agriculture on autumn staging Eurasian Cranes (*Grus grus*) in Estonia

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This paper explores the relation between the local numbers and distribution of autumn staging Eurasian Cranes (*Grus grus* Linn.) and agricultural land use during recent decades in Estonia. The analysis is based on the long-term monitoring data of staging cranes and the statistical data of land use in Estonia. We found that great changes in cropping area, as well as in crane numbers have taken place in Estonia since the 1960s. We also found a significant positive correlation between crane numbers and the cropping area of summer wheat, winter wheat, winter rye and all cereals together, and a negative correlation with the area of potatoes. Generally, arable land, particularly that used for growing cereals, has a great influence on the local numbers and distribution of staging cranes. Based on our findings, we predict that changes in the local numbers and distribution of Eurasian Cranes staging during their migration in Estonia and elsewhere will depend on changes in agricultural land use in staging areas, rather than on the size of the breeding population. As about 10 percent of the European Eurasian Crane population stop over in Estonia during the autumn migration, the country has an important role to play in the protection of the species.

Key-words: Eurasian Crane, staging cranes, crane protection, land use change, agricultural policy, Estonia

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Introduction

The Eurasian Crane is distributed in Eurasia from latitude 69°N to latitude 40°N and from longitude 6°E to longitude 165°E (Cramp and Simmons 1980, Prange 1989, Meine and Archibald 1996). Its recent European breeding population is about 110,000 pairs (Prange 2003, BirdLife International 2004). Breeding populations have increased across most of Europe. In Estonia the breeding population of the Eurasian Crane has increased from about 300 pairs in 1970 up to 6800 pairs in 2006 (Leito et al. 2003, 2006, Leito pers. comm.). Eurasian Cranes are omnivores, mostly feeding carnivorously during breeding and herbivorously during the migration and wintering periods (Cramp and Simmons 1980, Prange 1989, Díaz et al. 1996, Avilés et al. 2002). The composition of their diet depends on the season and local foraging opportunities. In Estonia, the main feeding habitats for cranes in the autumn are fields of different cereals and mowed grasslands (Leito et al. 2006).

During the period from the 1960s to the 2000s, the numbers of cranes that stopped over during the autumn migration has increased to a greater or lesser extent in all of the most important staging areas in Europe. The growth and relative importance of staging sites have, however, varied greatly from one year to another (Lundin 2005). In Estonia, the total number of staging cranes rose continuously during the 1960s and 1970s and has stabilised in the period from the 1980s to 2000s. At the beginning of the 1960s, up to 5000 cranes, and in the 1980s and 1990s, between 20,000 and 30,000 cranes were counted (Leito et al. 2006). All together, the Eurasian Crane population migrating on the West-European migratory route has increased from about 40,000 to 150,000 birds, and the number of cranes migrating on the Baltic-Hungarian route has increased from about 30,000 to 90,000 during the past 30 years (Prange 1999, 2003, Lundin 2005). Cranes breeding in Estonia use all the Eurasian Crane migratory routes in Europe, but are most numerous on the West-European and Baltic-Hungarian migratory routes (Leito et al. 2006).

Although Eurasian Crane numbers have increased substantially throughout Europe during recent decades, its breeding range has not yet reached the former distribution range, and its population has not yet recovered to the level that preceded its decline. In the list of Species of European Conservation Concern (SPECs) the Eurasian Crane is listed in SPEC category 2 (a species whose global populations are concentrated in Europe and which has an Unfavourable Conservation Status in Europe) (BirdLife International 2004).

Another aspect of the concentration of cranes on arable land during migration and wintering is that, if present in large numbers, cranes may cause serious damage to crops. There are two possible ways to solve this problem – to compensate the damages and/or create artificial feeding fields for cranes, and both schemes are in use (Koskinen et al. 2003, Lundin 2005, Nowald 2005, Petit and Couzi 2005). In Estonia, on the basis of the Fauna Protection and Use Act, the Regulation on Procedure and Methodology for Assessment of Damage Caused by Protected Animals or Birds on Migration was implemented in 1994. The guiding principle of the Regulation is to compensate the actual crop damage caused by animals and birds on migration, including that of staging cranes.

Many different policy measures, including agri-environment schemes, have been implemented across Europe, mostly addressing water, biodiversity, and landscape protection. In the late 1990s about 20% of farmland in the European Union (EU) was covered by national agri-environmental programmes (OECD 2003, Bayliss et al. 2005, Carey et al. 2005, Herzog 2005). Evaluation of their effects on biodiversity is, however, difficult from a methodological standpoint, and the existing studies are often controversial. More research on agri-environment schemes, other policy measures, and farming is needed in order to assess and improve their actual influence on different species, including cranes (Alonso et al. 1987, 1994, Prange 1999, Alonso et al. 2003, Bayliss et al. 2005, Lundin 2005).

We conducted a thorough survey of the numbers and distribution of Eurasian Cranes staging during the autumn migration in Estonia in relation to changes in agricultural policy and land use, particularly changes in cropping areas. The main goal of this study was to analyse the long-term dynamics of

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agricultural land use and numbers of autumn staging cranes in Estonia in order to assess whether and how agricultural practice affects the local numbers and distribution of staging cranes and to ascertain the importance of agricultural policy and land use for migrating birds.

Material and methods

Crane censuses

The autumn staging of the Eurasian Crane has been monitored in the Matsalu area since 1961, on Hiiumaa Island since 1982 and throughout Estonia since 1983. In Matsalu and Hiiumaa, censuses have been carried out almost every year. Total crane counts were conducted in the years 1983, 1994, 1999, 2000 and 2003. In these years the total counts were carried out in all the sites in Estonia where cranes stay during the autumn. Censuses were performed using the standard method developed in Estonia (Keskpaik et al. 1986). The census period lasts two weeks, from the middle to the end of September, with a central counting day on the weekend in the second half of September. According to this method, cranes are counted at the roosting sites during the flight from feeding site to roosting site in the evening. If the evening census is not successful due to bad weather (fog or heavy rain), the census is repeated the next morning.

Agricultural land use

The data on area and yields of winter rye, winter wheat, summer wheat, barley, oats, potato and hay, and for the total area of cereals and for all crops together in Hiiumaa and Läänemaa Counties and for the whole of Estonia from 1965–2005 were collected from the archive of the Estonian Farmers' Union and from the Statistical Office of Estonia (2006). Data on crop yields were not used for further analyses because the data was not complete for every year. The area characteristics of cultivated crops that we used should be even better than crop yield data, because cranes feeding on fields, apart from eating germinated grains or the leaves of sprouted cereals, also eat invertebrates, amphibians and small mammals living in the fields. In this way, the food source for cranes on arable land contains both the crops cultivated there and the accompanying edible small animals, and our results reflect the effect of the available food complex on crane distribution and numbers.

In Hiiumaa County (area 1019 km^2), the cultivated areas of crops coincide with the feeding area of cranes, since the birds feed on all fields throughout the island. The crane staging population of the Matsalu region is spread over about 2000 km^2, which makes up about 84% of the territory of Läänemaa County (area 2394 km^2). Consequently the agricultural indicators for the whole of Läänemaa County are representative for the Matsalu crane population. The total land area of Estonia without Lake Peipsi and Lake Võrtsjärv is 43,428 km^2 (Maansoo 2001), and agricultural land currently makes up about 20.5% (8890 km^2) of this (Statistical Office of Estonia 2002).

Data analysis

The coefficient of variation (*CV*) was calculated in order to demonstrate the temporal variation of crane numbers and cropping area. Ordination of cropping areas of different crops according to year in Hiiumaa, Matsalu and in Estonia as a whole were analysed using principal component analysis (PCA). The Spearman rank order correlation coefficient (r_s) was used to examine the relationship between crane numbers and cropping area in the same year in the Hiiumaa and Matsalu areas over the period 1965−2004. The Mann-Kendall test (*MK*) was used to find the presence of monotonic trends in time series. Locally weighted regression (LOWESS) was used to illustrate trends in the time series data of crane numbers. Correlations and LOWESS were calculated using the computer programme Statistica.

Results

Crane numbers and distribution

During the period 1982–2005, the number of autumn staging cranes on Hiiumaa has varied between 960 in 1998 and 4230 in 1993 ($CV = 1.69$, $n = 21$ years) (Fig. 1). Four different periods in crane numbers can be distinguished on the basis of figure 1: (1) relatively stable numbers at a low level in the 1980s; (2) population growth with a peak in numbers at the beginning of the 1990s; (3) a rapid decrease in numbers until 1998, and (4) relatively stable numbers from 1999–2005, remaining at the same level as in the 1980s.

During the period 1961–2005 the number of staging cranes in Matsalu has varied between 700 in 1996 and 21,500 in 1994 ($CV = 2.41$, $n = 30$ years) (Fig. 2). Three main periods can be distinguished for Matsalu on the basis of figure 2: (1) a substantial population growth from 1965–1983, (2) a period with fluctuating numbers at a level of about 10,000 cranes, and (3) a new growth period during the last decade since 1995. There is a significant positive trend in staging crane abundance over the whole study period 1961–2005 ($MK = 2.66$, $p < 0.01$, $n = 30$ years).

Based on total counts during the period 1983–2003, the number of autumn staging Eurasian Cranes in Estonia has varied between 18,000 in 1997 and 30,000 in 1994 ($CV = 0.51$, $n = 5$ years). The total number of cranes has fluctuated without any visible trend (Fig. 3). A total of 51 crane staging sites in 8 concentration areas were established in the 1980s, and 61 staging sites in 10 concentration areas in the 1990s. The main concentration areas of staging cranes are located in the western, south-eastern and northern part of Estonia (Fig. 4). All of the most important staging sites are situated in areas of large fields close to wetland. The most important gathering area has been Western Estonia, including the islands of Hiiumaa and Saaremaa, where, in different years between 1983 and 2003, 72% and 87% ($n = 5$) of all cranes were counted. The relative share of all staging cranes in Matsalu has been 48–71% ($n = 5$) during the period 1983–2003. This

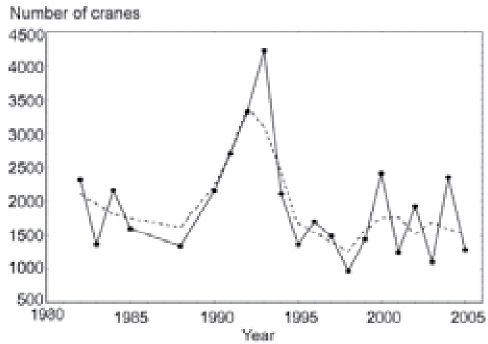

Fig. 1. Temporal dynamics of crane numbers on the island of Hiiumaa. The solid line represents the actual counted crane number and the dashed line reflects robust locally weighted regressions (LOWESS) over time.

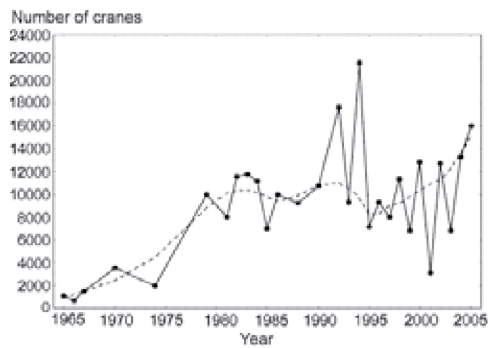

Fig. 2. Temporal dynamics of crane numbers in Matsalu. The solid line represents the actual counted crane number and the dashed line reflects robust locally weighted regressions (LOWESS) over time.

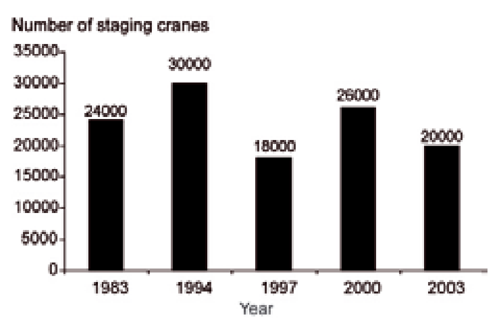

Fig. 3. Total numbers of autumn staging Eurasian Cranes counted in Estonia, 1983–2003 (After Leito et al. 2006).

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area is characterized by a large shallow-water sea bay and large fields in the surroundings. There is a significant positive correlation between the number of cranes staging in Hiiumaa and in Matsalu (r_s = 0.47, p < 0.05, n = 21) and between Matsalu and Estonia as a whole (r_s = 0.90, p < 0.05, n = 5).

Relationships between crane numbers and cropping area

During the period 1965–2004, the total area of cropland in Estonia has varied from 259,248 ha in 2002 to 444,223 ha in 1980 (CV = 0.55, n = 40) (Fig. 5a). The dynamics of the total area of cropland in Läänemaa County, Hiiumaa County and in the whole of Estonia has been similar (Fig. 5a, b). Five main periods in the total area of field crops can be distinguished for Estonia on the basis of figure 5: (1) the growth in total area in the period 1965–1976, (2) a relatively stable total area in the period 1977–1992, (3) a rapid decrease in the period 1993–1996, (4) a new increase and stabilisation on a lower level at the end of the 1990s, and (5) stabilisation in total area of field crops at a new level close to that of the 1960s. There is no significant linear trend in total area of field crops in Hiiumaa County (MK = 0.32, p > 0.05, n = 49), Läänemaa County (MK = 1.14, p > 0.05, n = 40) and in Estonia as a whole (MK =

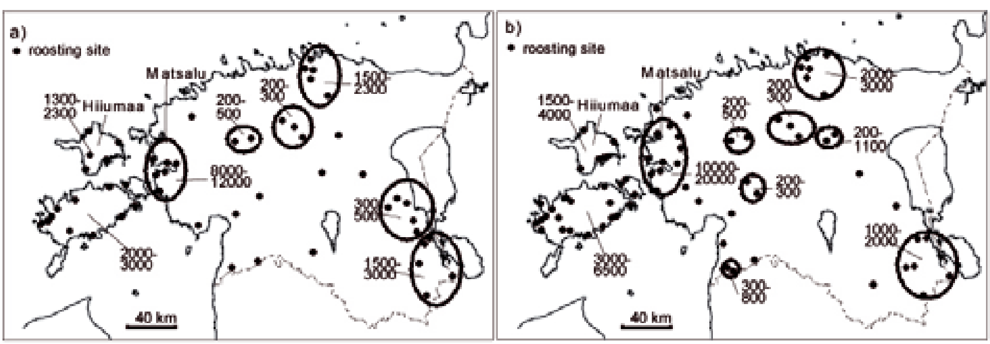

Fig. 4. Distribution of autumn staging Eurasian Cranes in Estonia in the 1980s (a) and 1990s (b).

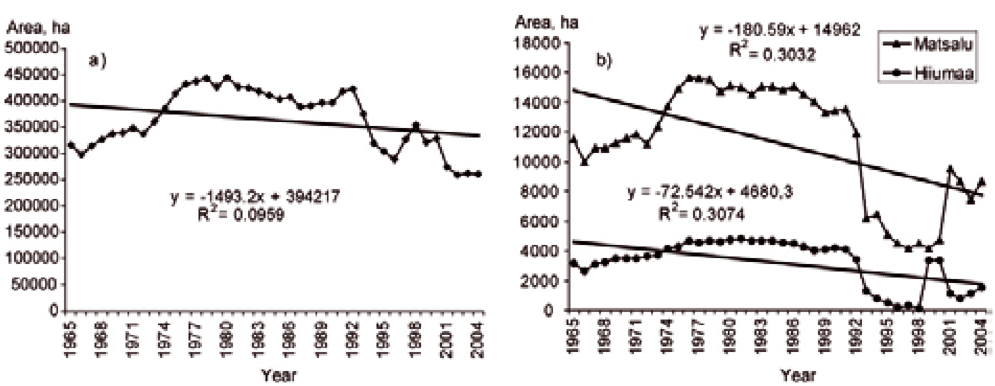

Fig. 5. Total cropping area in the whole of Estonia (a), and in Hiiu and in Lääne Counties separately (b) from 1965–2004. The trends are not significant (MK test, p > 0.05, n = 40).

0.47, $p > 0.05$, $n = 40$) over the whole study period 1965–2004.

The PCA analysis of the total area of cropland in Estonia indicated great changes in the relative share of different field crops in the period 1965–1990: the share of oats and potato had decreased and the share of cultivated grassland had increased (Fig. 6a). The total cropping area of all cereals together, except for summer wheat has decreased since the 1990s. In the Matsalu area (Läänemaa county) and on Hiiumaa (Hiiumaa county) the changes have been different compared to Estonia as a whole. In Matsalu the relative share of different field crops has been stable during the period 1965–1990, and has changed substantially in the period 1991–1995, when the cropping area of all cereals together decreased. Since 1996 the area of cultivated grassland and summer wheat has decreased (Fig. 6b). On Hiiuma

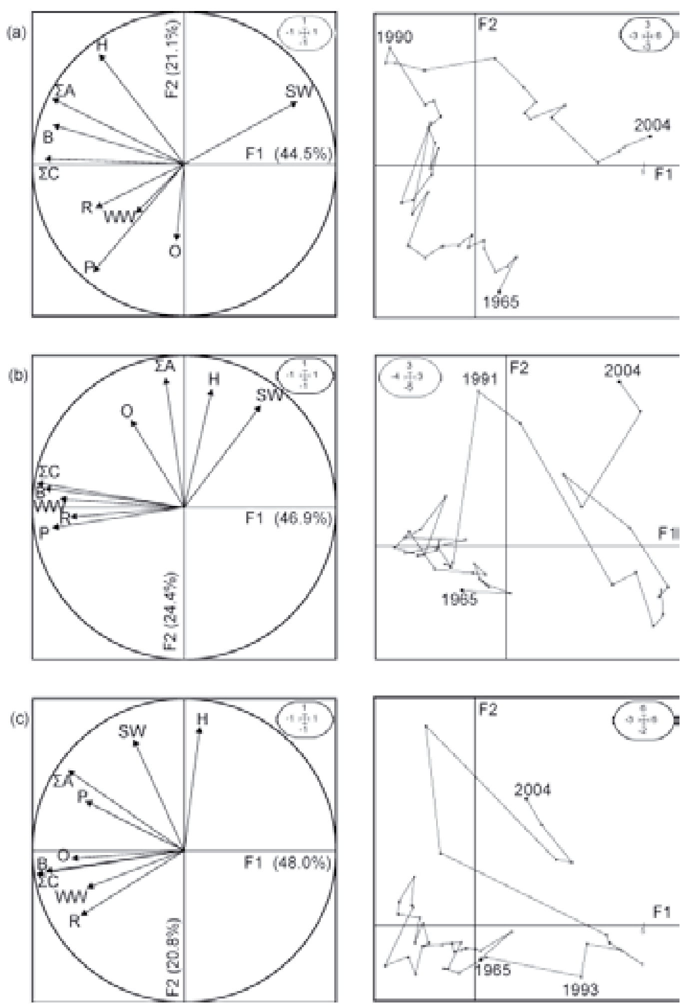

Fig. 6. Results of Principal Component Analysis (PCA) based on the correlation matrix of agricultural land use data. The plots on the left side illustrate the correlation of the agricultural variables (arrows) with the first two axes of (F1×F2) of the PCA. The plots on the right side illustrate the temporal dynamics of consecutive years with respect to the first two principal components: (a) Whole of Estonia, (b) Lääne County (c) Hiiu County. Abbreviations: R – winter rye, WW – winter wheat, SW – summer wheat, B – barley, O – oats, ΣC – total cropping area of cereals, P – potato, H – cultivated grassland, ΣA – total area of arable land.

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the changes have been similar to Matsalu during the period 1965–1990, but in the period 2001–2004 the structure of cropland on Hiiumaa has been more stable (Fig. 6c).

We found a significant positive correlation between the number of staging cranes and the cropping area of winter rye, winter wheat, summer wheat and all cereals combined, and a negative correlation between crane numbers and the cropping area of potato (Table 1). The correlation was strongest with the cropping area of winter rye (r_s = 0.58, $p < 0.05$, $n = 21$) and winter wheat (r_s = 0.58, $p < 0.05$, $n = 21$) on Hiiumaa, and with the cropping area of all cereals combined in Matsalu (r_s = 0.56, $p < 0.05$, $n = 28$). No statistically significant correlation between crane numbers and the area of cultivated grasslands was found in either Matsalu or Hiiumaa.

Table 1. Significant Spearman correlation coefficients (r_s) between the cropping area and the number of staging cranes in the Hiiumaa (n = 21 years) and Matsalu (n = 28 years) areas.

Locality	Crop	r_s	p-value
Hiiumaa	Winter rye	0.58	< 0.05
Hiiumaa	Winter wheat	0.58	< 0.05
Matsalu	Summer wheat	0.47	< 0.05
Matsalu	Potato	-0.41	< 0.05
Matsalu	All cereals together	0.56	< 0.05

Discussion

We found that the total numbers of Eurasian Cranes staging in Estonia during the autumn migration rose rapidly in the 1960s and 1970s, and stabilized in the 1980s. It is evident, that one reason for the increase in total numbers of autumn staging cranes in Estonia since the 1960s should be the overall growth in crane numbers breeding and passing through (Leito et al. 2006). Unfortunately, we cannot correlate directly the size of the local breeding population and the autumn population because we do not know the proportion of local birds in the autumn counts. On the basis of colour banding and radio tracking we know only that cranes from the local population and from Finland are mixed in Estonia during the autumn migration (Lundin 2005, Leito et al. 2006).

The Eurasian Crane breeding populations and the numbers of autumn staging cranes increased simultaneously up to the 1980s; after that the rise in staging crane numbers stopped, although the breeding population has continued to grow up to the present day (Leito et al. 2003, 2006). This difference is most likely related to agricultural land use changes. We found that staging crane numbers were positively correlated with cropping area of cereals and negatively with the extent of potato fields. It was predictable that the strongest relationship was between crane numbers and the area of winter rye and wheat. This is because, with these crops cranes can feed on newly sown fields and on green crops during one autumn, and on germinated fields and stubbles during the next. In this way the cranes exploit the same fields over a long period during two autumn seasons. Perhaps this is also one of the reasons why the correlation with barley fields was not as strong and was statistically insignificant compared to winter cereals. Breeding cranes are very fond of feeding in germinated barley fields and of picking the grains (Leito et al. 2006), but by the time most migrating cranes arrive in Estonia, the majority of fields have already been harvested, and the birds can only utilize the stubble fields. For that reason barley fields can be used by staging cranes mostly as stubble fields and only during one season. Only local breeding birds and very few early migrants can use germinated barley fields for feeding.

There is a similar but not exactly identical situation with summer wheat in Estonia. Whereas the barley harvest already begins in early August, summer wheat only ripens from late August. Early-arriving cranes will to a certain extent also feed on the germinated summer wheat fields. Another reason why the correlation between crane abundance and the cropping area of rye and wheat was stronger than with barley may be the greater abundance of other food for cranes, such as insects, amphibians and small mammals on these fields. This aspect of crane food has not been studied in Estonia, but some studies from wintering grounds in Spain demonstrate the importance for cranes of additional food other than the main crop they feed on in their habitat (Reinecke and Krapu, 1986; Díaz et al., 1995; Guzmán et al., 1999; Avilés et al., 2002). Visually, rye and wheat fields seems to be much richer in additional food for cranes, if only because of the higher stand of the germinated crop and stubble compared to barley.

In the summer non-breeding cranes in Estonia locally feed in potato fields (Leito et al. 2006). They pick new potatoes right from the furrow. In the autumn the fields are harvested and only a few potatoes or other food remain for the cranes. The negative correlation between autumn crane abundance and the extent of potato fields can be explained mostly by the conflicting relationship with the area of other crops more important for feeding cranes, i.e. when the area of potato crops in a locality increases, the cropping area of cereals and the potential food source there decrease respectively.

The main driving forces in present day rural landscapes in Estonia are land reforms, political campaigns, land amelioration, concentration and intensification of agricultural production. Most probably, newly cultivated lands, especially large field systems, have been the most important factor for the rise and development of autumn staging crane assemblies in Estonia during the 1950s and 1960s (Leito et al. 2006). During the period from the 1960s to the late 1980s collective farms throughout Estonia were becoming fewer and bigger, and their land use was getting more concentrated to the farm centres and to newly-cultivated fields. By this time large tracts of farm land in the periphery had already been abandoned. These processes were direct results of State policy – to support intensive land use in large collective farms.

After Estonia regained her independence in 1991, the transition from a centralised system to a market economy, land reform, and the privatisation of state farms began (Alanen 1999). This resulted in profound changes in agriculture and related land use. The agricultural reform was carried out during the period from 1992 to1997 – collective farms were dissolved and re-organised mostly into joint-stock enterprises or private farms. The changed trade conditions caused vast changes in land-use – large fields were abandoned and the total area of cereals declined. The cultivation area of agricultural crops decreased and the area of unused arable land increased 20-fold by 1999 (Sepp and Hiiemäe 2003). According to expert estimates, in 1999 there were 330–350 thousand hectares of unused arable land in Estonia. (27–31% of all arable land). In accordance with the data of the Statistical Office of Estonia (2002), in 2001, 32.6% of arable land went unused. Some recent changes in the distribution of cranes on migratory routes and in wintering areas are clearly caused by changes in agricultural land use, depending on EU agricultural policy (Alonso et al. 1987, Alonso et al.1994, 2003). More recently, before and after accession to the EU in 2004, rural development programs have been established that encourage the re-cultivation of abandoned agricultural land.

Besides the food sources discussed above, the comprehensive protection of the species and its habitats, and the warming of the climate are probably related to changes (increase) in the distribution and numbers of cranes staging in Estonia in the autumn. The hunting of Eurasian Cranes is currently prohibited in all European countries and illegal hunting is significantly reduced (Meine and Archibald 1996, BirdLife International 2004). In Estonia, the hunting of the Eurasian Crane was prohibited in 1958 and illegal hunting is insignificant (Leito et al. 2006). Also, the establishment of many new protected areas in places where the Eurasian Crane has been breeding and resting has probably contributed to the population growth in Estonia and elsewhere. The proportion of protected areas in Estonia has risen from 4% in 1970 to 12% in 1999 and 16% in 2006 of the total area of the country (Fammler et al. 2000, Leito et al. 2007). Recently, more cranes are wintering in France and Germany, particularly because of climate warming, and their migratory route is shorter compared to earlier times (Cramp and Simmons 1980, Alonso et al. 2003, Lundin 2005). A shorter migration route decreases energy-expenditure and risks during migration, which contribute to a higher survival rate (Berthold 1993, Leito et al. 2003).

In general, we found that migrating crane patterns in Estonia are related to cropping patterns.

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And, as the agricultural land use on a county level has changed greatly, depending on agricultural policy, we conclude that it too has had a great impact on migrating cranes. Agri-environmental measures play an important role in decreasing the area of abandoned land and increasing the area of grasslands and cereals. Also direct subsidies to cereal growers have helped enlarge the area of fields that are suitable for the autumn staging of cranes. The resulting abundance of food for staging cranes has led to an increase in crane numbers.

The changes in the pattern of land-use in Hiiumaa and Matsalu have been similar to the general trends in Estonia, except for the last years, when the structure of arable land use in Hiiumaa and Matsalu has been more stable compared to Estonia as a whole.

Acknowledgements. We are very grateful to all those who have taken part in crane censuses in Estonia over the past several years. The Estonian Science Foundation (grants no. 3735 and 6007), the Estonian Ministry of the Environment and the Estonian Environmental Investments Centre provided financial assistance for the performance of Eurasian Crane studies. We are also thankful to Alexander Harding and Ilmar Part for linguistic editing.

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III

Kaasik, A., Raet, J., Sepp, K., Leito A., Kuusemets, V., 2008

LAND USE CHANGES ON HIIUMAA ISLAND
(NORTH-WESTERN ESTONIA) IN THE
LAST FIFTY YEARS

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Land use changes on Hiiumaa Island (north-western Estonia) in the last fifty years

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Abstract

Significant social, economic and political changes of the last 50 years have altered Estonian rural landscapes. The paper analyses changes in land use intensity and landscape patterns in the two main agricultural regions (altogether 267 km^2) of Hiiumaa, the second largest island of Estonia. The spatial analysis of land use patterns, which was based on decoded aerial photos (ortophoto maps) from 1956 and 1984, the digital Basic Map of Estonia from 1998, fieldworks in 2004/5 and performed in GIS software MapInfo, showed an overall decrease in agricultural land (esp. in account of natural and wooded grasslands) and gradual increase in forested land. The greatest change in land use pattern took place between 1956 and 1984. By 1984, the traditional and extremely diverse patchy mosaic landscape pattern of 1956 had become much more simplified and polarised as a result of collectivisation, land reclamation and wider use of industrial methods in agriculture. Since the beginning of the 1990s remarkable changes in agricultural land use intensity, a sharp decline by the end of the 1990s and a slight revival by 2004/5, have not yet caused any significant changes in landscape patterns.
Keywords: rural landscapes, land use change, Hiiumaa.

1 Introduction

There are several studies about general land use change in Estonia during the 20th century. Mander and Palang have brought out the main tendencies like a considerable increase in forests, decrease in agricultural lands and continuing

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decline in grasslands; and the main socio-economic factors that have led to the land use change [1, 2]. Some of the driving forces like land reforms (in 1919, 1940, 1947 and 1989), deportations and collectivization (in 1940s) and formation of the Soviet border zone along the coastline are specific for Estonia or for the former Soviet (the so-called post-Soviet) countries only, but other more general factors, related to improvement of agricultural practices, urbanisation etc., as well as their impacts are widely spread all over the world. Concentration of agriculture, marginalization, land amelioration, use of bigger machines etc., have reshaped rural landscapes in many countries and have brought about the loss of valuable semi-natural land use types (e.g. wooded and coastal meadows etc.) that have formed as a result of long-term traditional agricultural practices [3, 4, 5].

Unlike previous Estonian studies (e.g. Palang et al [2], Koppa [6], Tomson [7]), the current land use analysis is focused on the last 50 years, specially on agricultural landscapes and is based on large-scale cartographical materials (1:10 000). The study area embraces two agricultural regions in Hiiumaa: Hellamaa (200 km^2) in the north-eastern and Vanamõisa (~67 km^2) in the southern part of the island, fig. 1. In Hiiumaa the share of agricultural land has drastically decreased from more than 65% in 1939 to less than 25% in 1992 [8] and nowadays most of the agricultural land use in Hiiumaa is concentrated in these regions.

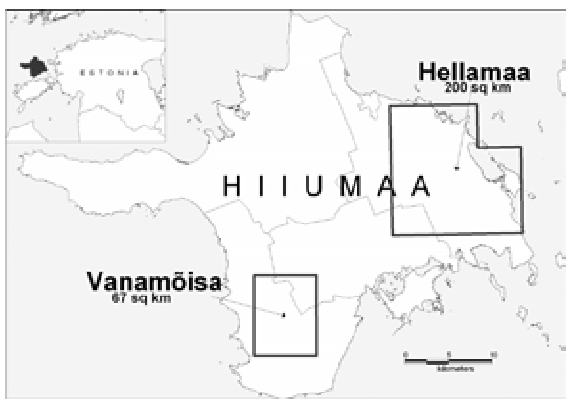


Figure 1: Location of study areas.

Due to relative isolation and poor preconditions for agriculture (young and mostly stony and thin soils on limestone), the landscape changes in Hiiumaa have been slower and the land use had retained more or less a traditional character by the middle of 1950s. So it was presumed that the first available aerial images from 1956 reflect the diverse land use pattern that had been developed by old traditional extensive agricultural practices during the first independence period (1918–1939) of Estonia. Different aspects of historical land use of Hiiumaa have been studied by many authors, e.g. Hellström has analysed the development of the farming landscapes, settlements etc. in the south-eastern part of the island since the middle of the 19th century [9].

2 Methology

The landscape analysis was based on decoded aerial photos (ortophoto maps) from 1956 and 1984. The large-scale ortophoto maps (1: 10 000) were scanned and the land use patterns of 1956 and 1984 were digitized in GIS software MapInfo. The state of the present land use was identified by the digital Basic Map of Estonia (1:10 000) based on aerial images from 1998. In 1956, a slightly more detailed classification of land use was used and the additional land use types like pastures and fallowed fields were distinguished, but roughly the classification has remained quite similar. The main distinguished land use types were: fields, fallow fields (in 1956), grasslands, pastures (in 1956), bushes, sparse woodlands, young forests, forests, clearings and wind falls, mires, lakes, rivers and ditches, reed-beds, partly vegetated sandy and stony areas, ruderal areas, peat production areas, stone heaps, courtyards, buildings, roads, parks and greeneries, graveyards, quarries, power-line corridors and sea. In order to analyse the successive transformations between the land use categories, the land use types were digitized as precisely as possible, concerning grasslands, pastures and mires the existence of bushes, trees or both was defined. By field survey carried out in summer 2004/5, the actual land use, i.e. the fields still in use, long-term and short-term fallows, were identified. In many cases, the agricultural areas had been abandoned (not ploughed, moved or pastured) till the beginning of the 1990s (i.e. since the collapse of collective farming). The land use patterns were analysed and the first statistics and landscape metrics were calculated in MapInfo. In addition, large-scale (1: 10 000) land use maps of former collective and state farms, all compiled in 1983, were used.

3 Findings

The land use pattern of 1956 largely reflects the patchy mosaic landscape pattern of the previous independence period (1918–1939) which was characterised by comparatively small fields, grasslands and woodlots. By 1939 in Hiiumaa 71% of all land was owned or used by peasants, the average farm size was 23.4 ha [9] and the share of agricultural land (more than 65%) was one of the highest in Estonia [8]. The landscape pattern of 1984 is much more simplified and polarised as a result of amelioration, collectivisation and a wider use of industrial methods in agriculture. By that time the former extremely diverse patchy type of landscape was re-organised and replaced by large fields and extensive forests, fig. 2.

The landscape patchiness has decreased steeply in both study areas from 1956 to 1984, fig 3(A). The total patchiness, with artificial objects (roads, buildings, etc.) excluded, follows the same tendency/trend from 15667 patches in 1956 to 8607 patches in 1984, fig. 3(B). The Edge Density (ED, m/ha) calculated for the terrestrial areas declined from 188.1 in 1956 to 137.7 in 1984. By 1998 the landscape pattern and patchiness (esp. concerning fields and cultivated grasslands and pastures) has remained almost the same as in 1984, only the majority of fields and grasslands were fallowed as the agricultural land use has attained probably the lowest level by that time. According to the Estonian Rural

Figure 2: Examples of land use changes.

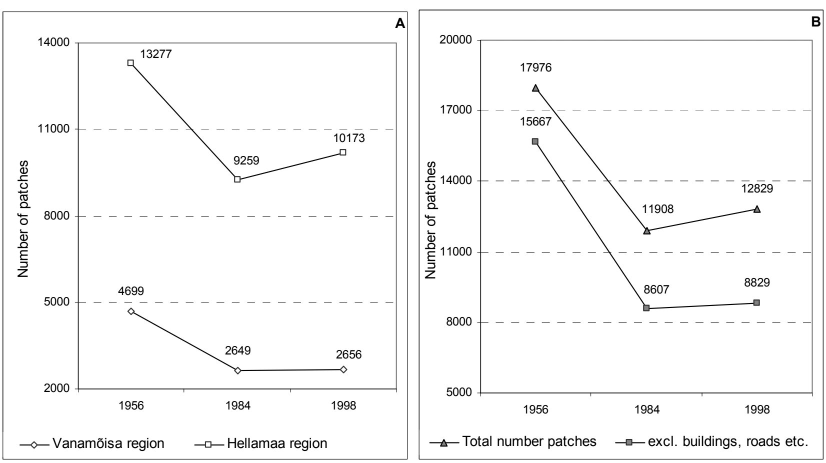


Figure 3: Change in landscape patchiness.

Development Plan of 2004-2006 (ERDP), the use of agricultural land in Hiiumaa has decreased by up to 50% from 1993 to 2001 [10].

Another tendency was the overall decrease in agricultural land (mainly in account of grasslands) and increase in forests, fig. 4. The land use changes were quite similar in both study areas, but in Hellamaa region the rise in forests was

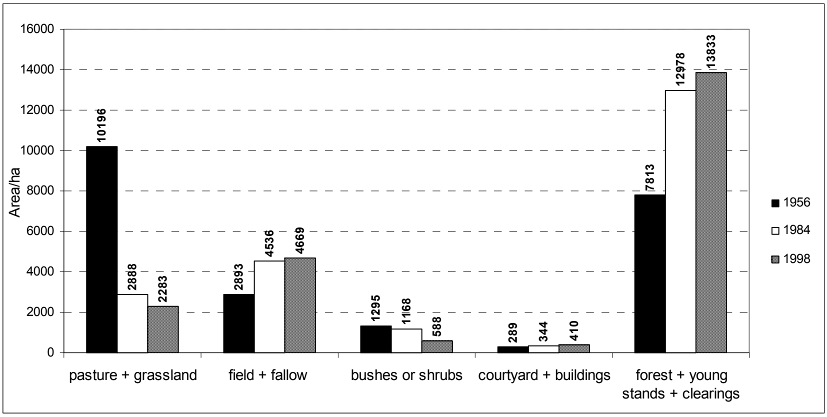


Figure 4: Changed land use.

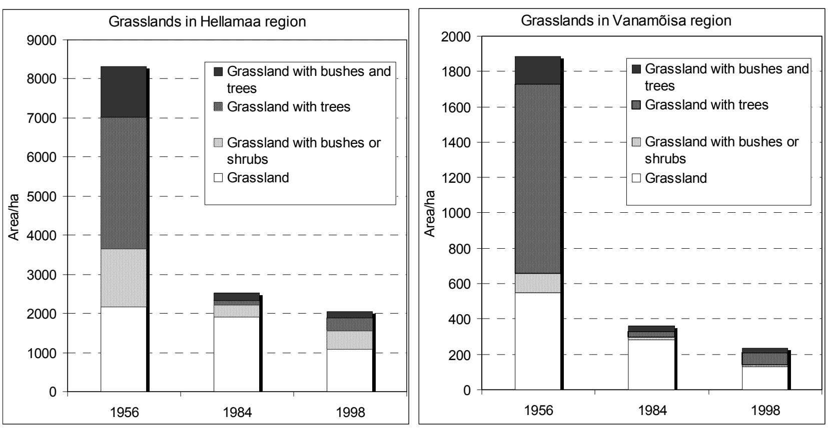


Figure 5: Decline in grasslands.

much more significant, the area of forests increased by more than twice during the observed period.

The share of grasslands and pastures has decreased significantly especially in account of natural and wooded meadows, i.e. grasslands with bushes and/or trees by our classification, fig 5. Most of the grasslands have afforested or overgrown with shrubs while others have been turned into fields and cultivated grasslands or other land use categories. The total area of pastures in 1956 was 4505 ha. By 1984 more than half of the pastures (50.4%) had transformed into forests and 13.4% into fields. The transformation of open grasslands (total area 5706 ha in 1956) was quite similar, by 1984 43.7% of the grasslands were transformed into forests and 24.4% into fields, fig. 6. From 1984 to 1998, the transformation of previous pastures and grasslands continued mainly by successive

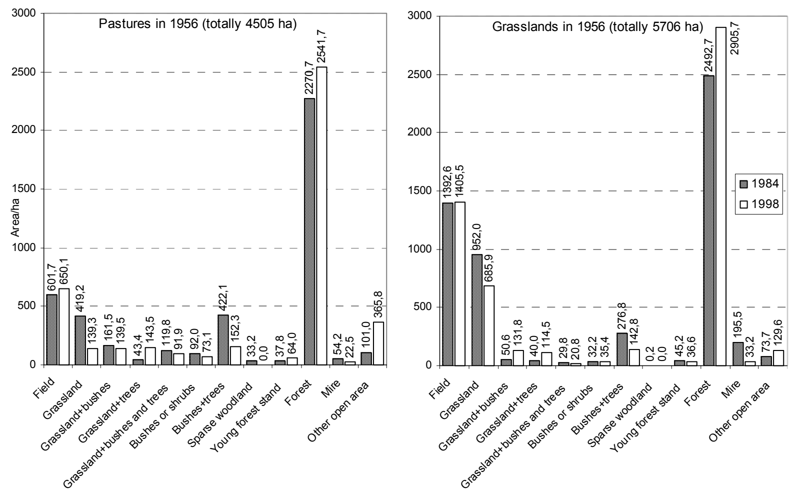
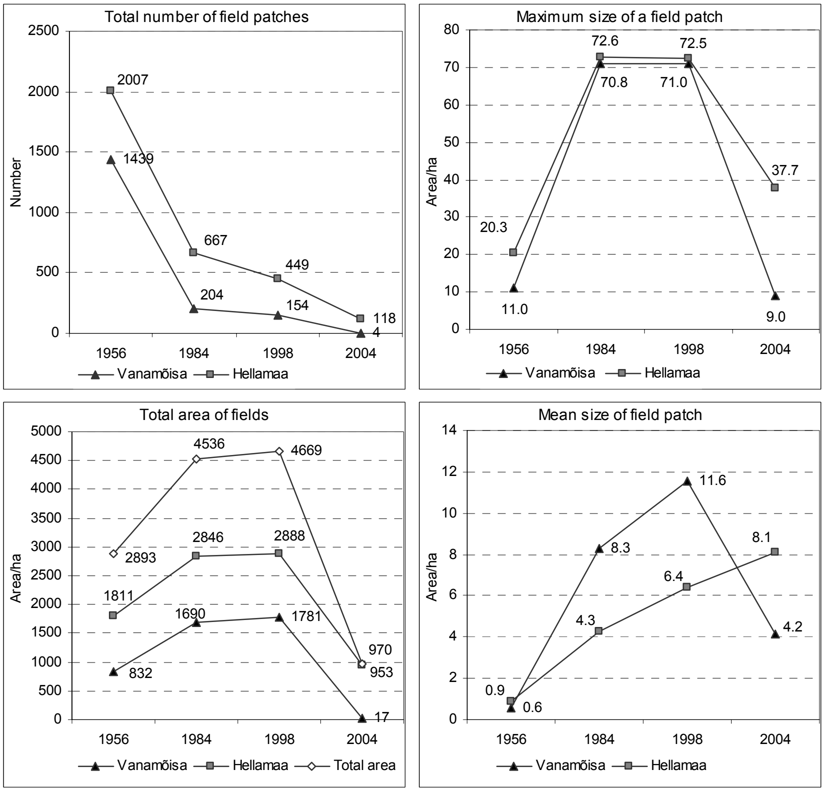


Figure 6: Transformation of pastures and grasslands from 1956 to 1998.

transformations, i.e. by gradual overgrowing with bushes and trees. By 1998 more than 56% of the pastures and about 51% of grasslands had transformed into forests and only 3.1% of the previous pastures and 12% of grasslands had maintained and were used as open grasslands.

The basic statistics about fields show a significant change between 1956 and 1984, when the total number of field patches had decreased in both regions (in Hellamaa region about 3 times and in Vanamõisa region 7 times, respectively). At the same time, the mean and maximum size of a field patch as well as the total area of fields had increased. Between 1984 and 1998, the above-mentioned tendencies leading to a homogeneity of fields' structure continued, but at a substantially slower rate. In 1998, the majority of fields that were included in calculations had been fallowed for a longer or shorter time, but were still classified as fields, fig. 7. By that time, the agricultural land use had attained probably the lowest level [11]. The expectations that land re-privatisation and formation of small-scale private farms since the beginning of the 1990s would restore the diverse and well-managed rural scenery, that was characteristic of the previous independence period, did not come true by the end of the 1990s. In fact, a great portion of the privatised agricultural areas were fallowed by that time. According to ERDP, the use of agricultural land in Hiiumaa has decreased by up to 50% from 1993 to 2001 [10]. In 2004, a slight revival in agricultural activity was observed, probably as a result of the enforcement of the EU agricultural subsidiary system. By that time some long-term fallowed fields and grasslands were freshly put into agricultural use again. A spatial analysis of the fields of 1956, 1998 and 2004 by soil fertility (expressed in soil quality points and identified on a large-scale digital Soil Map of Estonia) shows that the average soil fertility of the fields was quite uniform during the whole period: 35.4 in 1956, 37.7 in 1998 and 37.0 in 2004, respectively.



* 2004 - only cultivated field patches are taken into account

Figure 7: Some basic statistics about arable lands.

According to the land use maps of state and collective farms (compiled in 1983), in Hiiumaa the amelioration of agricultural lands was started in 1963 and was carried on till the beginning of the 1980s, although the major portion of reclamation works (altogether 92.3% of all reclaimed areas) were implemented between 1968 and 1979, fig. 8. Cyclic decrease and increase of lands embraced

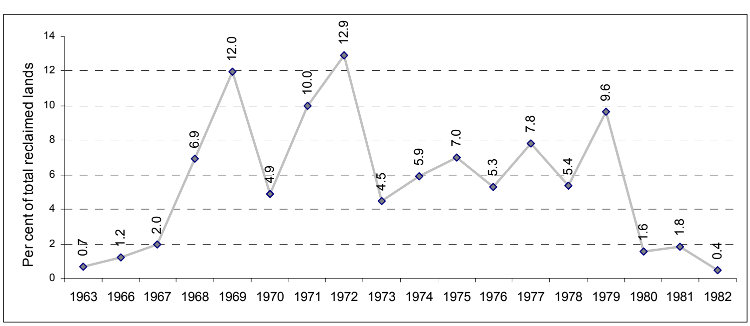


Figure 8: Amelioration of agricultural lands.

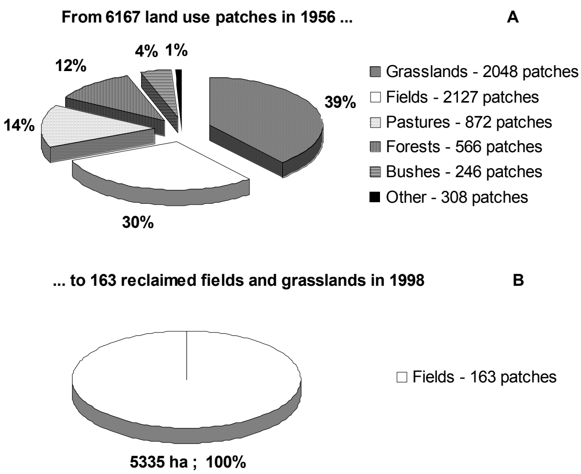


Figure 9: Simplification of landscape structure due to land reclamation.

to amelioration process can be related with state politics and five-year plans in its implementation.

In order to analyze the impact of land reclamation on the landscape structure, the larger ameliorated fields and cultivated grasslands (in total 163 fields) with a minimum size of 1.72 ha, maximum size of 236.3 ha, mean size of 32.7 ha and total area of 5335 ha were selected, fig 3(B). The spatial analysis inside the large ameliorated areas showed a significant simplification of land use structure. The 163 large fields that have formed as a result of land reclamation have transformed from 6167 tiny land use patches in 1956. The fields have been transformed mainly in account of grasslands and pastures (53%, 2920 patches), former fields (30%, 2127 patches), forests (12%, 566 patches) and bushes (4%), fig 3(A). Other land use types (roads, courtyards, mires and water bodies, etc.) formed altogether 1%, whereby e.g. only 4 ha of mires (9 patches) and 7 ha of water bodies (lakes, ponds and streams; 26 patches) have disappeared due to land reclamation of the fields.

4 Discussion

The significantly changed land use and landscape pattern of Hiiumaa should influence the species composition, population numbers and distribution of species (incl. birds) living or staging in certain land use types. The main interest in the large-scale spatial land use data of the last 50 years was induced by an ambition to compare and combine it with the available biodiversity data, esp. with the existing reliable bird census data from the same period. Hiiumaa (but mainly its agricultural region) is an important place for many waterfowl and other species. Most impressive numbers of birds can be seen during the spring and autumn migration. For instance, the cranes and geese feed on their migration route practically only on agricultural land parcels (cereal fields, cultivated grasslands and pastures). It is supposed that their breeding success, as well as the species' general state and multitude are, to a high degree, dependent on these agricultural resources. For now, correlations between the numbers of the Eurasian Crane (*Grus grus*) and statistical data of cereal crops production has been already confirmed [12].

5 Conclusions

(1) In Hiiumaa the agricultural landscapes have changed considerably during the last 50 years.
(2) The land use pattern has simplified and polarised as a result of marginalisation, land reclamation, collectivisation and intensification of farming. The greatest decline in total patchiness and in edge density (ED) took place between 1956 and 1984.
(3) The area of agricultural lands has decreased in account of grasslands in total by about 43% and the area of forests has increased by about 44% from 1956 to 1998. The share of grasslands has decreased significantly especially in account of natural types of grasslands.
(4) By 1998 more than half of the pastures and grasslands of 1956 were transformed into forests.
(5) Greatest decrease in field patchiness and increase in the total area of fields took place between 1956 and 1984. By 1998 the field patchiness and the area of fields had remained almost the same as in 1984, but actually the majority of fields were fallowed by that time. A slight revival in agricultural activity occurred in 2004, probably as a result of the enforcement of the EU agricultural subsidiary system. The remarkable changes in agricultural land use intensity since the beginning of the 1990s have not yet caused any considerable changes in landscape patterns. As regards soil fertility, there was no significant difference between the used and abandoned fields in 2004.
(6) The major portion of reclamation works were done between 1968 and 1979. The ameliorated fields have been transformed mainly in account of grasslands and pastures (53%), former fields (30%), forests (12%) and bushes (4%). The 163 larger fields in 1998 have been transformed from more than 6000 land use patches in 1956.

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IV

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Soil biota indicators for monitoring the Estonian agri-environmental programme

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Abstract

The mandatory implementation of agri-environmental programmes (AEP) during the accession to the EU was a major policy development in the countries of Central and Eastern Europe. In Estonia the development of the AEP began in 1997 and the implementation of the AEP in pilot areas was launched in 2001. By this time, evaluation and monitoring methodologies of AEP had been elaborated. This paper summarises the Estonian AEP applied in 2001–2003 and presents selected indicators of biodiversity. The main focus is on the analysis of selected indicators of soil biota. Soil bioindicators (abundance, diversity, and ecological composition of earthworm communities and hydrolytical activity of the microbial community) were measured in 2001 and 2002. Two pilot areas of the AEP (Palamuse, representing a municipality with intensive agriculture, and Kihelkonna–Lümanda, representing extensive agriculture) and two reference areas where the AEP was not implemented (Mustjala and Saare municipalities) were investigated. Ten farms in both pilot areas and five in both reference areas were studied. There were differences in the abundance and number of earthworm species between Palamuse (intensive agriculture) and Saaremaa (extensive) pilot areas, but these were not statistically significant. However, the differences in the hydrolytical activity of the microbial community between Palamuse and Saaremaa pilot areas were statistically significant ($p < 0.05$). It is concluded that soil bioindicators are suitable for monitoring human pressure as well as the effects of AEPs. AEPs may increase the abundance and diversity of earthworms, decrease the dominance of *A. caliginosa*, and increase the activity of the microbial community.

Keywords: Monitoring; Agri-environmental programme; Biodiversity indicators; Soil biota; Earthworms

1. Introduction

The development and implementation of agri-environmental programmes (AEP) is a legal requirement for all EU Member States since the agri-environment regulation, Council Regulation No. (EEC) 2078/92, accompanied the reforms of the common agricultural policy in 1992. The mandatory implementation of AEPs during the accession to the EU was a major policy development in the countries of Central and Eastern Europe (including Estonia), and is likely to have a significant impact on farm management practices and the patterns of land use. In Estonia, the development of

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the AEP began at the end of 1997 with the "Agri-Environmental Programmes in Central and Eastern Europe" (MATRA) project (Avalon Foundation, 1998). Preparations continued under the PHARE projects "Development of the Agri-environmental Scheme in Estonia" and "Action Plan for Implementation of the Agri-environmental Programme for Estonia, N°ES 9620.01.01". The structure of the Estonian agri-environmental support scheme, the measures and the requirements for eligibility, and the administration and training system were elaborated during these projects, and a plan was drafted for the implementation of the scheme. The main activities and measures with their intended positive effect on biodiversity and landscape protection are presented in Table 1. For example, the aim of the "Environmentally Friendly Production Scheme" is to encourage the use of environmental planning by farmers and to reduce the risk of water pollution by nitrogen, while maintaining and increasing soil fertility. Farmers are required to have a Nutrient Management Plan and a Crop Rotation Plan. The total application of nitrogen (mineral fertiliser and manure/slurry) must not exceed an average of 170 kg/ha of cultivated area, and the total application of nitrogen as mineral fertiliser must not exceed 100 kg/ha of cultivated area. The crop rotation must meet the following requirements: at least 15% of the cultivated area must be under legumes or a mixture of legumes and grass species, and cereals must not be grown on the same field for more than three subsequent years.

Estonia started to implement agri-environmental support in 2000, when assistance for organic farming and Estonian cattle breeding was made available nationwide. In the following year, nationwide assistance was granted, in addition, to the production of endangered crop varieties. The Ministry of the Environment began to support the management of semi-natural habitats in protected areas. At the same time, several additional agri-environmental measures were initiated in three pilot areas. The specific objectives of this pilot project for agri-environmental measures were:

- testing the practical implementation of the AEP in the Estonian context;
- evaluating the effectiveness of the proposed AEP;
- refining the management prescriptions of the AEP in order to better meet the objectives of the scheme;
- gaining an indication of the average AEP payments per farm;
- demonstrating and promoting the concept of the AEP in Estonia amongst farmers, policy makers, politicians and the general public;
- developing a control, monitoring and evaluation system;
- elaborating evaluation indicators for the AEP and testing them in the pilot areas.

2. Material and methods

2.1. Approaches for the selection of indicators in the Estonian AEP

2.1.1. Biodiversity indicators

Indicators are partial and imperfect reflections of reality. For the evaluation of the agri-environmental

Table 1
Pilot agri-environmental measures in Estonia (2001–2003)

Scheme	Main activities and measures favourable for biodiversity and landscape protection
Environmentally friendly Management Scheme (EMS)	Minimum requirements with respect to crop rotation
	Restricted nitrogen fertilisation
	Establishment of non-cultivated field margins
	Limitations of field size
	Maintenance of landscape elements, semi-natural and natural habitats
Supplementary Measures Scheme (SMS)	Restoration and management of semi-natural habitats
	Planting of hedges
	Restoration and management of stone walls
	Creation of ponds and wetlands
	Preservation of endangered local breeds, organic farming
Abandoned Land Scheme (ALS)	Shrub clearance and annual mowing starting from a certain date
Training and Demonstration Scheme	Training and demonstration

programme, we need many indicators, since there are several goals in the management and protection of agricultural landscapes. Indicators achieve political effectiveness only if they really contribute to policy development, which would suggest that indicators are powerful enough to support the formulations of political goals. To have such a strong policy impact, the suite of indicators must give correct and complete information, which allows us to assess whether a system evolves towards sustainability or not (Piorr, 2003). Indicators suitable for one function (habitat function, landscape aesthetic, etc.) may be totally inappropriate for others; effective indicators have a format that is designed with an explicit target group in mind (Braat, 1991; Notter and Liljelund, 1993).

The selection of biodiversity and landscape indicators is difficult, because both are complex and value-laden systems (OECD, 2001). Moreover, the processes and interactions within and between ecosystems are very complex. Despite the large number of articles on biodiversity indicators (e.g. CEC, 2000; OECD, 2001, 2002; Braband et al., 2003; Büchs, 2003a, 2003b; Herrmann et al., 2003; Hoffmann and Greef, 2003) there is no such thing as a universal set of biodiversity indicators. Each agri-environmental programme needs its own set of indicators depending on the objectives that should be achieved.

While selecting indicators it is important to adopt a hierarchical approach, linking the indicators to their respective level of analysis, such as field, farm and landscape. Some indicators can have relevance only at specific scales of analysis, while others can be used at different spatial levels. For instance, the indicator 'diversity of the scenery' has significance at the level of 'landscape', whereas the indicator 'length of field boundaries' is meaningful both at the level of 'field' and of 'agro-ecosystem'.

In our research we adopted the Drivers-Pressures-State-Impact-Responses (DPSIR) framework to promote the development of agri-environmental programmes. This is the framework proposed by international organisations such as the Organisation for Economic Co-operation and Development (OECD) and the European Environmental Agency (EEA) to integrate the complex information concerning the environment into the decision making process (OECD, 1999, 2000, 2001, 2002). The DPSIR framework is used for developing and integrating indicators for measuring the state of the environment, the driving forces (e.g. human socio-economic activities) that exert pressures on natural resources, and the impacts on the environment resulting from pressures. A group of Estonian experts adopted the DPSIR framework to the local situations and estimated the impact of pressures on biodiversity state indicators (Table 2).

In order to interpret results, biodiversity monitoring needs benchmarks. National indicators describing environmental functions were identified, and the corresponding Environmental Minimum Requirement (EMR) values to allow their performance were assessed (see also Table 5). Baseline values were identified for each biodiversity indicator in each study area and were compared to their actual values. This allows us to define whether the agricultural use of an area is within its "ecological sustainability" and whether it maintains or enhances its environmental functions or, on the contrary, impairs them. In the current article we focus on soil biota indicators.

2.1.2. Indicators of soil biota

Soil biota play a key role in the functioning of soils, and some parameters can indicate the impact of human activities on the soil. There is evidence that measures of the size and activity of the soil biota communities, e.g. earthworm numbers, microbial biomass, soil respiration and soil enzyme activity, have considerable potential as early indicators of soil degradation or improvement (Haynes and Tregurtha, 1999). The use of soil macro-fauna – and earthworms in particular – as bioindicators relies on their prominent place in the community of soil organisms and their function in promoting processes that are considered to be linked to soil health (Doube and Schmidt, 1997). They are sensitive and efficient indicators of changes in soil health and for assessing the effects of agri-environmental measures (Haynes and Tregurtha, 1999; Paoletti, 1999; Büchs, 2003a; Schloter et al., 2003).

Earthworms possess a number of qualities that animals being used for bio-monitoring of terrestrial ecosystems need; they are large, numerous, easy to sample, easily identifiable, widely distributed, and relatively immobile; they are in full contact with the substrate in which they live and they consume large

Table 2
Relation between pressure and national indicators of biodiversity in pilot areas

State indicators	Pressure indicators								
	Nutrient management		Livestock density		Crop rotation	Pesticide use	Land management		
	High amount of mineral fertilisers	High amount of manure	High livestock density	Low livestock density	No planned crop sequence	High amount of pesticides	Low landscape maintenance	Abandonment	Simplified landscape structure and large field
Abundance and composition of plant species in fields and field edges	−2	−1	−1		−2	−3		−2	−2
Number and diversity of carabids (*Carabidae*) in fields and field edges	−2	−1	−1	−1	−2	−2		−2	−2
Number and diversity of soil earthworm (*Lumbricidae*) communities	−2	−2	−1	−1	−1	−2		−/+1	
Functional structure and hydrolytical activity of soil microorganisms	−2	−2	−1	−1	−1	−2		−/+1	
Presence of protected species (communities) in agricultural landscape	−2	−1	−2	−1		−2		−/+1	−2
Presence of indicator species in agricultural landscape (birds, bumblebees)	−2	−1		−2	−2	−2			−2
Diversity of cultivated crops					−3				−2
Share of managed semi-natural grasslands in the total area of semi-natural grasslands				−3				−3	
Share of natural area	−1	−1				−1		+2	
Average field size					−2		−3		
Total length of field margins					−2				
Share of managed linear landscape elements from total linear landscape elements							−3	−2	
Variability of different land cover types					−1			−1	−2

	No impact	3	High impact
1	Low impact	+	Positive impact
2	Significant impact	−	Negative impact

The table shows the hypothetical estimated impact of pressure indicators for agricultural soil use on state indicators for biodiversity.

volumes of this substrate (Edwards and Bohlen, 1996). They are traditionally considered to be convenient indicators of land use and soil fertility (Graefe, 1999; Paoletti, 1999). According to Lavelle and Spain (2001), the regional abundance of earthworms and the relative importance of the different ecological categories are determined by large-scale climatic factors (mainly temperature and rainfall) as well as by their phylogenetic and biogeographical histories together with regional parameters such as vegetation type and soil characteristics.

Earthworm community composition and diversity allows us to infer on soil microclimatic and nutritional conditions. According to different authors (Edwards, 1983; Paoletti, 1999; Curry et al., 2002), earthworm abundance and diversity in cultivated land are generally lower than those found in undisturbed habitats; agricultural activities such as ploughing, various tillage operations, fertilising and the application of pesticides strongly affect these animals. Decreases in earthworm abundance can be directly attributed to injuries caused by cultivation practices, or indirectly to habitat disruption and reduction in food supply, as well as high predation during and after tillage operations. Paoletti (1999) suggests the use of biomass, species number, and ecological categories (e.g. epigaeic, endogaeic and anecique) as key indication parameters in agro-ecosystems.

In addition, soil microbial community parameters have been widely used in the monitoring of soil quality (Anderson, 2003; Büchs, 2003a; Schloter et al., 2003). The soil microbial community is a component of the terrestrial ecosystem that assures the degradation of organic residues and the biogeochemical cycling of minerals. As mentioned by Beylich and Graefe (2002), the microbial activity of the soil influences its O_2 concentration and consequently the survival of earthworms.

The soil microbial properties (microbial biomass, enzyme activities) provide a reliable tool for the estimation of early changes in the dynamics and distribution of soil microbial processes within soil profiles (Kandeler et al., 1999). The validation of the discriminant function by other data sets may well yield a means of predicting the temporal dynamics of the response of soil microbial communities and their functional diversity to different management practices used in agro-ecosystems.

2.2. Study areas

In order to effectively pilot the national agri-environmental programme (AEP), two pilot areas were selected according to the following criteria:

1. contrasting areas with respect to natural conditions which shape the landscape;
2. contrasting areas with respect to agricultural activity (including farmers' attitudes), biodiversity and landscape interest;
3. areas with the following common characteristics necessary for successful project implementation:

- easily defined, with clear boundaries (e.g. by the administrative boundaries of local municipalities), and of a manageable size;
- availability of secondary data (i.e. to make best use of limited resources for data collection);
- local political support/approval of the project and AEP in general, co-operative local partners;
- clear values to protect and agri-environmental problems to solve.

Taking into account all above-mentioned selection criteria, Palamuse municipality in Jõgeva County and Kihelkonna and Lümanda municipalities in Saare county (hence forth Saaremaa pilot area) were selected in 2001 as pilot areas for testing the national AEP and the proposed agri-environmental indicators (Fig. 1) (Table 3).

Saaremaa pilot area represents an example of extensive agriculture in Estonia. The agriculture of the Saaremaa study area was also extensive during the Soviet period (before 1991) and the use of agrochemicals was much lower than the Estonian average. Since the end of the Soviet era it has further decreased and at present, most of the farmers in the Saaremaa pilot area use very little or no mineral fertilisers, and pesticide use has dropped to almost zero. None of the monitored farms were using any pesticides. The average size of fields was 2 ha. The main soil type according to FAO classification (FAO-UNESCO, 1994) is pebble rendzinas (*Rendzic Leptosols*). In Palamuse, which is a typical area of intensive agriculture, the use of fertilisers is much higher (up to 170 kg/ha of NPK); pesticides are used mainly for cereal cultivation with average quantities between 0.8 (mixed crop) and 3.8 l/ha (oat). The average field size in Palamuse was

Fig. 1. Location of AEP pilot areas (Kihelkonna–Lümanda and Palamuse) and reference study areas (Saare, Mustjala).

Table 3
Application of selection criteria in the pilot areas

Selection criteria	Palamuse community	Kihelkonna and Lümanda communities (Saaremaa pilot area)
Size	Total area: 21,607 ha; total agricultural land: 10,286 ha	Total area: 44,543 ha; total agricultural land: 8841 ha
Natural factors	Situated in Upper Estonia[a] Hillocky landscape. More continental eastern climate	Situated in Lower Estonia[a] on Silurian limestone. Semi-maritime climate
Agricultural activity and biodiversity/landscape	An area of intensive agricultural production with relatively fertile soils. Unique (drumlin area) heritage landscape value	An area of extensive agricultural production on thin pebble rendzinas. Encompasses valuable (high biodiversity) semi-natural habitats
Project administration factors	Co-operative local actors present: Community government, Jõgeva County government	Co-operative local actors present: Community government, Viidumäe Nature Protection Reserve
Availability of secondary data sources	Good	Good
Local political support/approval	Community government was very interested to have its community chosen for a pilot area and they recognised the need for agri-environment measures	Community governments were very interested to have their communities chosen for a pilot area and they recognised the need for agri-environment measures

[a] Estonia is divided into Upper and Lower Estonia. After the retreat of the glacier, the waters of ice lakes or of the Baltic Sea inundated Lower Estonia during the last 10,000 years in Holocene. Lower Estonia is more marshy, more densely wooded, and flatter than Upper Estonia, which has been untouched by flooding from glacial lakes and the sea in Holocene.

4.8 ha. Typical brown soils (*Calcaric Cambisols*) and pseudopodzolic (*Podzoluvisols*) soils are dominant. In both pilot areas, 10 farms who had joined the AEP and had implemented the measure of "Environmentally Friendly Production Scheme" were selected. In each farm a cereal field was selected for monitoring. For both pilot areas reference areas (Mustjala for Saaremaa and Saare for Palamuse) were selected and in both areas, five farms not participating in the AEP were monitored. In these farms, fields of similar size and with the same crops as the AEP study farms were chosen.

2.3. Measurements

In both pilot test areas of the AEP, soil biotic indicators were measured in 2001 and 2002. Monitoring parameters of the earthworm populations were: abundance of earthworms, number of species, biomass per m^2, mean fresh body biomass, ecological composition of community, dominancy; additionally hydrolytical activity of microbial community was determined. Soil moisture content and acidity, as the most important limiting factors for earthworms, were also measured.

In all study fields, five randomly selected soil blocks of 50 cm × 50 cm × 40 cm in the centre of the field were studied by the hand-sorting method (Satchell, 1969; Meyer, 1996); samples were washed and weighed and species were identified according to Graff (1953) and Timm (1999). The mean number of individuals per m^2 of soil surface and standard error (S.E.), as well as the ecological composition and dominancy of communities, were calculated. In all composite soil samples, gravimetric moisture content (at 105 °C), pH (KCl), total N (Kjeldahl) and organic matter content (at 360 °C) were measured.

The total activity of the microbial community was measured using the fluorescein diacetate method, which estimates the activity of dehydrogenase enzymes in a composite sample (Schnürer and Rosswall, 1982). All data was analyzed using the dispersion analysis of Kruskall–Wallis.

3. Results and discussion

The climatic conditions in 2001 were suitable for earthworm communities. The summer (June–September) mean air temperatures were close to normal for these regions. In the Saaremaa pilot area the summer mean temperature is 14.5 °C; in 2001 it was 14.2 °C. In Palamuse the summer mean temperature is 13.3 °C; in 2001 it was 14.4 °C. Soil moisture conditions were optimal for earthworms between August and September, when average soil moisture at depth 0–40 cm was 15.6% in the Palamuse pilot area, 14.9% in Saare municipality, 21.5% in Saaremaa and 18.1% in Mustjala municipality.

There were statistically significant differences in some soil characteristics (nitrogen concentration, soil pH). Saaremaa soil pH was 7.2 ± 0.2; Palamuse soil pH was 5.9 ± 0.2. Organic matter content and total nitrogen concentration were also higher in Saaremaa soils compared to Palamuse soils (organic matter content 7.5 ± 2.0 and 2.8 ± 0.3%, total nitrogen content 0.47 ± 0.14 and 0.14 ± 0.02%, respectively).

Differences in abundance and number of earthworm species between AEP pilot areas and their reference areas (Lümanda–Kihelkonna and Mustjala; Palamuse – Saare) did not exist. In both Saaremaa and Palamuse test areas, six earthworm species were found. Only the species tolerant to agricultural activities (one epigeic, three endogeic, two anecic) were discovered and more sensible species were not found. The dominant species in communities was *Aporrectodea caliginosa* (81 and 89%, respectively, of all individuals). The abundance of earthworms was 32.0–224.0 m^{-2}, 1–5 species, in the Palamuse area. The abundance of earthworms was 0–614.0 m^{-2}, 0–5 species, in the Saaremaa area. There were differences between Palamuse and Saaremaa pilot areas, but these were not statistically significant (Table 4). The abundance and number of species was higher in Saaremaa; no differences were found in earthworm biomass and the ecological composition of communities between the two areas. Eighty one percentage of the individuals in Saaremaa and 89% of the individuals in Palamuse belonged to the earthworm species most tolerant to human impact, *Aporrectodea caliginosa*; although it is not statistically significant, this difference gives an indication of the higher intensity (use of fertilisers, lack of crop rotation) of agricultural activity in the Palamuse area prior to the implementations of the AEP. We expect the number of individuals of the dominant species *A. caliginosa* to decrease and some other species (*Aporrectodea rosea*,

Table 4
Characteristics of earthworm and microbial communities (mean value ± S.E.)

Parameter	Palamuse (intensive agriculture)	Saare (reference area to intensive agriculture)	Kihelkonna/Lümanda (extensive agriculture)	Mustjala (reference area to extensive agriculture)
N, individuals per m^2	117.1 ± 18.6	111.4 ± 24.2	149.3 ± 58.6	150.7 ± 55.6
S, number of species per m^2	3.1 ± 1.0	3.33 ± 1.0	3.3 ± 1.3	3.0 ± 1.5
M, g per m^2	45.5 ± 11.4	27.0 ± 8.4	45.4 ± 21.2	82.9 ± 54.2
Body mass of individual, g	0.37 ± 0.03	0.27 ± 0.07	0.39 ± 0.04	0.66 ± 0.03
Dominancy, %	88.2 ± 4.1	90.3 ± 3.5	80.4 ± 5.0	82.9 ± 13.3
Epigaeic earthworms, %	1.0 ± 0.7	0	4.3 ± 2.0	0.8 ± 1.4
Endogaeic earthworms, %	83.2 ± 9.5	95.4 ± 2.3	69.0 ± 11.9	87.7 ± 28.0
Anecique earthworms, %	5.8 ± 2.0	6.7 ± 2.6	6.7 ± 2.6	11.5 ± 8.2
Hydrolytical activity, OD per g dry soil	0.54 ± 0.05	0.57 ± 0.11	0.87 ± 0.09	0.74 ± 0.05

Table 5
Environmental Minimum Requirements (EMR) of monitored soil parameters in pilot areas

Parameter	Mean value	Minimum value	Maximum value	EMR[a]
Abundance of earthworm communities, individuals per 1 m^2, Palamuse	96	32	224	82
Abundance of earthworm communities, individuals per 1 m^2, Saaremaa	104	0	614	65
Number of earthworm species on 1 m^2, Palamuse		1	5	2
Number of earthworm species on 1 m^2, Saaremaa		0	5	2
Hydrolytical activity of microbial community, OD per 1 g dry soil, Palamuse	0.622	0.375	1.022	0.618
Hydrolytical activity of microbial community, OD per 1 g dry soil, Saaremaa	0.756	0.446	1.128	0.751

[a] Environmental Minimum Requirement (EMR) values are calculated on the basis of the national environmental monitoring data from the period 1995–2001, separately for different landscape (soil) regions. EMR = mean value − S.D. (for both areas $n = 60$).

Lumbricus rubellus, *Lumbricus terrestris*, etc.) to be found after a prolonged application of AEP.

The hydrolytical activity of the microbial community was significantly higher ($p < 0.05$) in Saaremaa than in Palamuse (0.87 ± 0.09 and 0.54 ± 0.05 OD (Optical Density, according to Schnürer and Rosswall, 1982) per gram dry soil, respectively). The differences between AEP pilot areas and their reference areas (Lümanda–Kihelkonna: 0.87 ± 0.09; Mustjala: 0.74 ± 0.05) were not statistically significant.

We have derived Environmental Minimum Requirements (EMR) for Saaremaa and Palamuse areas using the mean values and standard deviations of parameters of earthworm and microbial communities (Table 5). All calculated soil indicators were above EMRs, which indicates the stability of soil ecosystems.

4. Conclusions

The pilot project provided valuable experience in testing agri-environmental measures and indicators of biodiversity for evaluating and refining new developments in the Agri-environmental Programme in Estonia.

The set of state indicators for biodiversity (Table 2) provides a comprehensive framework to assess the level of human pressure on different categories of agricultural land. Within this framework, measures of the size and activity of soil biota, e.g. abundance, diversity and ecological composition of earthworm communities, hydrolytical activity of the microbial community, have considerable potential as early indicators of soil degradation or improvement and could be applied for monitoring the effects of AEP. There were differences in the abundance and number of earthworm species between Palamuse (intensive) and Saaremaa (extensive) pilot areas, but these were not statistically significant. However, the differences in the hydrolytical activity of the microbial community between Palamuse and Saaremaa pilot areas were statistically significant ($p < 0.05$). Two years is a too short period to evaluate the impact of AEP measures on soil biota. The collected data will be a baseline for future evaluations of the effects of different AEP measures.

The possible results of the application of AE measures are an increase in the abundance and diversity of earthworms, a decrease in the dominancy of *A. caliginosa*, an increase in the activity of the microbial community.

Comparisons between the Environmental Minimum Requirement (EMR) values of the chosen indicators and their actual values allow to assess whether agricultural use of an area is within the range considered ecologically sound and whether it maintains or enhances the area's environmental functions or, on the contrary, impairs them. The concept of Environmental Minimum Requirement is difficult to apply for some indicators, because scientific evidence is lacking. Still, wherever possible, it can be used to support the policy maker in defining quantified ecological objectives.

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Neighbourhood-defined approaches for integrating and designing landscape monitoring in Estonia

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Abstract

Landscape monitoring is a rapidly developing approach in the field of environmental science and management. In order to develop a sound landscape monitoring programme, key theoretical concepts and study objectives should be clearly stipulated, and the specific objects to be monitored, as well as the criteria for selecting study areas, hierarchical levels, and techniques of data collection and analysis should be identified. This paper describes the development and implementation of the Estonian monitoring programme for agricultural landscapes, conventional approaches for landscape monitoring, and by neighbourhood analysis, assesses how landscape features are covered by different complementary monitoring data and how the current pattern of monitoring networks represents the landscape features. A spatially explicit method of network design for monitoring and sampling strategies combines stratified and multi-scale agricultural landscape monitoring and uses neighbourhood analysis characterised by the nearest neighbourhood index and Ripley's *K*-function. Data for landscape analysis are obtained from landscape monitoring (three sets) and other complementary environmental monitoring sets, such as biodiversity, forest, soil, and water monitoring (11 sets). It is shown that several monitoring sets follow an approach that aims to achieve national geographical coverage, representing various landscape types. Small sets having less than 50 stations are biased and the networks are not statistically significant. Proportional stratified sampling requires fewer sites for large homogenous inland landscape districts. The concept of agricultural landscape monitoring was tested in pilot areas. The chosen multi-scale object-based methods provide a good overview of the level of human pressure on different categories of agricultural land. Results of the monitoring showed that the species composition and abundance of bio-indicators was, to a great degree, determined by landscape structure. A systematic approach focused on landscape classes helps to integrate the monitoring set as a whole and to achieve a coherent and efficient layout of monitoring sets for Estonia.

Keywords: Landscape monitoring; Neighbourhood analysis; Agricultural landscapes; Estonia

1. Introduction

Environmental, including landscape monitoring can be seen as a process by which we maintain an overview of the state of the environment. It provides essential data on the ways systems are changing and how rapidly. In addition, it provides essential feedback to management, so that we can adjust what we are doing and get the best information out of the system. In several countries a special scientific research programme on landscape monitoring has been established (O'Neill et al., 1994; Ihse, 1995; Winkler and Wrbka, 1995; Herzog et al., 2001), and in some countries landscape monitoring programmes have already been launched (Barr et al., 1993; Bunce et al., 1993; Fuller et al., 1993; Fuller and Brown, 1994; Howard et al., 1995; Roots and Saare, 1996; Ihse and Blom, 1999; Groom and Reed, 2001; Bailey and Herzog, 2004).

The first landscape monitoring programmes focused mostly on land cover aspects (Bunce, 1979). The need for objective information on land cover was recognised in Britain as early as the 1930s when Stamp (1962) implemented the Land Use Survey. Over recent years, landscape mapping and classification has evolved to become a highly sophisticated science with extensive use of satellite remote sensing data (Griffiths and Mather, 2000; Mücher et al., 2000). The exploration of the dynamics of landscape structural features and landscape compositional analysis are important topics in scientific research in many countries (Bailey and Herzog, 2004). The landscape monitoring methodologies have become more sophisticated, covering various landscape elements from biodiversity and vegetation,

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through the analysis of abiotic landscape components, such as soils, water systems, and landscape structure, to anthropogenic and cultural aspects, such as scenery and landscape aesthetics (Bunce, 1979; Gulinck et al., 1991; Barr et al., 1993; Brandt et al., 1994; Cherill et al., 1994; Fuller et al., 1994; O'Neill et al., 1994; Hulshoff, 1995; Winkler and Wrbka, 1995; Ihse, 1996; Seibel et al., 1997; Aaviksoo, 1998; Mücher et al., 2000; Dramstad et al., 2001; Herzog et al., 2001; Bastian et al., 2002; Brandt et al., 2002; OECD, 2002; Bailey and Herzog, 2004; Groom, 2004). Often, programmes of landscape monitoring are policy driven (Groom and Reed, 2001) or focus on specific values, i.e. the properties of intact landscape that provide services to society and that we wish to maintain (O'Neill et al., 1994). Values change as societies and their natural capital change (Haines-Young et al., 2003), and monitoring programmes are adapted and developed accordingly.

2. Scope and objectives

Many authors have emphasised that there are no readily available methodologies for landscape monitoring (O'Neill et al., 1994; Herzog et al., 2001; Groom, 2001, 2004). There are only a few standardised status reports on landscapes. For example, 3Q in Norway and LIM in Sweden elaborate a reporting standard for agricultural landscapes (Blom and Ihse, 2001; Fjellstad et al., 2001). There is, however, an evolving set of basic principles for designing a monitoring programme. Thus, when developing a landscape monitoring programme, one should first define the theoretical concept for monitoring, the objectives and objects to be monitored, and the criteria for selecting study areas. In addition, one should define optimal methods of data collection, acquisition, and analysis (use of landscape indicators, time series), followed by tests in pilot areas and applications of the methodology at a national level. In practice, every monitoring programme is unique, depending mostly on geographical coverage, landscape features, range of monitoring, available technology, and financial capacities.

Whereas some aspects of landscape, such as the structure or land cover, can be monitored through specifically designed landscape-monitoring programmes, often a number of other landscape elements, such as soil, habitat, and water are monitored through independent studies. In this paper we propose the integration of landscape monitoring using primarily the concepts of geocomplexes and neighbourhood within the framework of the Estonian national monitoring programme. A data set on landscape features, stressing neighbourhood relations, configuration, and coherence of the environmental monitoring networks for integrated landscape analysis is tested. We explore what dataset is provided by agricultural landscapes monitoring and what data could additionally be obtained from other environmental strata, and what spatial unit might be employed for interpolation of datasets.

2.1. Development of landscape monitoring in Estonia

In general, the dynamics of land use structure are an important indicator of socio-economical and political changes in society. Since 1991, the process of land reprivatisation in Estonia has been under-way. Over 200,000 farmer owners or their heirs are claiming back their land. The impact of land reform on landscape structure has been unpredictable. In 1992 the Agricultural Reform Act was passed. The purpose of the Agricultural Reform Act was the liquidation of collective and state farms (*kolkozes* and *sovkhozes*) and the transition to agriculture based on private ownership. Slow and incomplete privatisation and an inadequate rural policy have resulted in extensive land abandonment. This has created several environmental problems—a decrease in biodiversity and in the aesthetical value of the landscape, a rise in the distribution of weed seeds and the danger of fire. Taking this context and these problems into account, the main objectives of landscape monitoring programmes were defined as:

- To determine the landscape structure.
- To follow landscape changes and to predict future trends on the national level.
- To give statistics and an overview on the state of Estonia's landscapes.
- To obtain information enabling optimisation of the use of landscapes as a resource.
- To explain the relationships between landscape diversity indicators and other environmental characteristics (e.g. characteristics of the ecological status).
- To compile a comprehensive reference list on Estonian landscape diversity.

Since January 1994, a National Monitoring Programme has been implemented in Estonia under the supervision and coordination of the Ministry of the Environment. The main purpose of the programme is to monitor long-term and large-scale changes in the environment and thus identify the problems that call for operational measures or complementary studies in the future (Roots and Saare, 1996). A draft concept of the Estonian landscape monitoring programme was presented to the Estonian Ministry of the Environment in 1995 (Sepp and Kaasik, 1995). To develop the Estonian monitoring programme, experiences from other countries were examined. For example, "Landscape Monitoring and Assessment Research Plan" (O'Neill et al., 1994), "Countryside Survey 1990" (Barr et al., 1993; Bunce et al., 1993; Fuller et al., 1993; Fuller and Brown, 1994; Howard et al., 1995) and LIM-project in Sweden (Blom and Ihse, 2001) were assessed for the background, and aspects were incorporated into the Estonian plan.

The Estonian national landscape monitoring programme concept introduced four monitoring sub-programmes: agricultural landscapes, coastal landscapes, protected and valuable landscapes, and land cover (Sepp, 1999). Since 1996, three programmes (monitoring of protected and valuable landscapes and land cover monitoring were combined) have been implemented (Table 1). In developing a landscape monitoring programme, several aspects were considered, including: available technology (GIS and spatial database tools, satellite images, aerial photos); the objectives and structure of existing Estonian and European monitoring programmes; institutional and financial capacity; and the scientific principles of landscape ecology (Fig. 1).

Table 1
The current indicator set of landscape monitoring in Estonia

Name of sub-programme	Objectives of the programme	Monitoring method	Recorded parameters	Derived parameters
Monitoring of agricultural landscapes	To study changes in land cover types, and linear and point features of landscape structure. To explain the connection between landscape structure indicators and the characteristics of ecological status of agricultural landscapes	Black-and-white aerial photos from different time periods, satellite imageries, time series analysis, spatial statistics, field surveys and mapping land cover, linear and point features of landscape indicators of ecological state of landscape	Land cover, linear and point features of landscape, ecological state of landscape on bio-indicators (bumblebees, earthworms)	Edge index (m / ha); length of linear elements per ha of monitoring area (agricultural land); number of point elements per ha of monitoring area (agricultural land); number of patches per ha.
Monitoring of coastal landscapes	To identify the natural variability of habitat patches in coastal landscapes and to estimate the loss of and pace of fragmentation of habitats due to anthropogenic pressure	Field inventories	Vegetation type, land use class and ownership (type) for each patch within the site	Number of identified habitat types; number of identified land use classes; total number of patches; gamma-diversity index (Shannon); total length of patch; perimeter, indicators of human pressure
Monitoring of land cover	To study changes of land cover	Digital multi-spectral classification of Landsat Thematic Mapper or equivalent images to delineate physiognomic patches using ground truth information for land cover classes	Identification of land cover classes	Total number of classes; total number of separate patches; the total area of patches; mean patch size; maximum patch size; the number of patches per 10 000 ha, 100 ha; edge index (m / ha); neighbourhood index; shape index; total length of patch perimeter; density of patch perimeters (m / ha); density of patches (number/ha); average perimeter of patches; Shannon (γ) diversity index; contagion index

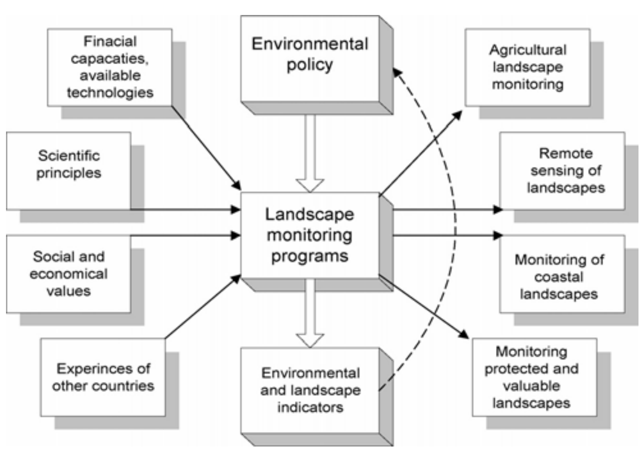

Fig. 1. The concept of the landscape monitoring programme in Estonia as applied here.

Table 2
Environmental monitoring programmes complementary to landscape monitoring in Estonia

Programme	No. of station	Main parameters	Frequency	Working scale	Size of test site	Representation and modelling techniques
Agricultural landscapes	18	Landscape structural elements, bioindicators	Every fifth year	1:5,000; 1:10,000	2 km × 2 km	Categorical mapping, landscape typologies, transects
Coastal landscapes	26	Landscape elements, plant diversity	Every fifth year	1:5,000	1 km × 0.5 km	Categorical mapping, typology of coastal habitats; historical mapping
Remote sensing of landscapes	7	Landscape cover classes, landscape structural indicators	Every fifth year	1:50,000; 1:10,000	Continues	Automated classification
Biodiversity: rare and threaten plant communities	144	Status and coverage of threatened plant communities (bogs, alvars, forests, meadows)	Every fifth year	1:1,000	50 m × 50 m	Local uniqueness, within landscape district, categorical mapping of land cover
Biodiversity: birds	110	Change in species composition of threatened, protected, and/or internationally important birds	Every fifth year	1:10,000	100 m transect up to 10 km × 10 km	Transect, major nesting sites
Biodiversity: rare plant species	225	Status of populations of rare plant species	Every fifth year	1:10,000	2 m × 2 m; 10 m × 10 m	Categorical mapping of land cover
Biodiversity: soil's biota	17	Earthworms, soil micro-organisms	Every fifth year	1:1,000	15 m	Categorical mapping of land cover and soil texture's typology
Forest monitoring	96	Temporal and spatial variations in forest conditions in relation to the occurrence of factors; interactions between the various components of forest ecosystems; pollutant and nutrient balance	12 per year	1:10,000; 1:1,000	0.25 ha	Plot at 16 km × 16 km grid; classification; generalised linear model, spatial autocorrelation; kriging
Soil monitoring	8	Quality of soil	Annually	1:10,000	50 m × 50 m	Kriging; categorical mapping of soil districts
Integrated monitoring	2	Small ecosystems to determine impacts and changes; geochemical analysis	12 per year	1:400	1 ha	Categorical mapping of land cover
Surface-water monitoring: rivers and lakes	114	Human impact of water use and quality, chemical and biological status; pollutant and nutrient balance; changes due to land use	12 per year	1:10,000	Water body	Classification; categorical mapping; spatial aggregation
Ground water monitoring	464	Human impact of water use and quality; quantitative and qualitative status	4 per year	1:10,000; 1:5,000	Groundwater body	Classification; hydrodynamic model; spatial aggregation; kriging
Air monitoring	26	The status of air pollution and the pollution load; deposition, pollutant balance; critical loads	On line	1:10,000	1 km × 1 km up to 50 km × 50 km	Dispersal model
Meteorological	59	Data on meteorology and hydrology	On line	1:10,000	1 km × 1 km	Dispersal model, kriging

Landscape has been attributed several different meanings and interpretations, and a single common understanding does not exist. When establishing a landscape monitoring system, it is essential that the landscape definition is suitable for the phenomenon and process under consideration and that regional context and spatial arrangement are taken into consideration (Bailey and Herzog, 2004). In developing the landscape monitoring programme in Estonia, the landscape was defined as a regional unit, or geo-complex (Arold, 1991). Landscapes were considered as dynamic material systems formed by the interaction of substances and processes within the geo-sphere. Their ingredients or components are interrelated with each other both in their development and their spatial location. Every landscape is inherently a geo-complex, in which a change in one component (land cover, vegetation, or the water regime, etc.) affects the whole complex.

A fundamental question in developing a monitoring programme is the selection of an approach for designing a set of monitoring areas. The monitoring network needs to be optimised in both spatial and temporal scales, aiming at the appropriate data density and quality and at efficient sampling strategies. Theoretically, a random monitoring network is the best way to exclude subjectivity and to give landscape features the opportunity to be chosen by chance (Bunce et al., 1996; Brandt et al., 2002; Bailey and Herzog, 2004). At the same time, random monitoring often requires a vast number of monitoring sites, depending on the selected monitoring variables, and is thus often too massive and quite expensive. Alternatively, a strategic approach using data collected for multiple purposes is often chosen. Large areas are subdivided into landscapes or eco-regions. For example in the UK, Spain, Flanders and Austria, national and regional monitoring systems for landscapes and land use are based on environmental strata (Winkler and Wrbka, 1995; Brandt et al., 2002). In this case, proportional samples of regions are used, with sample size relatively smaller for large homogeneous regions. Also, the procedure is more cost-effective, because large uniform areas require less sampling. In agricultural landscape monitoring in Estonia, a strategic approach in the selection of monitoring areas has been promoted. In selecting study areas, a representative distribution according to the Estonian landscape districts is assumed.

3. Methods

3.1. Methodology framework

The following questions arose: what potential relationships can be identified between various environmental and landscape districts and how can the measurement of landscape elements be calibrated to illuminate the relationship with other environmental datasets? We conducted two investigations: (1) a stratified topic-based selection exercise by neighbourhood analysis and (2) a multi-scale object-based monitoring of agricultural landscapes. In the first investigation, a selection key was developed by neighbourhood and identified landscape units. As additional supporting sources of landscape data, the Estonian national environmental monitoring set of 11 monitoring themes, which incorporates 1316 monitoring stations and reports a total of 227 parameters, was used. The most relevant parts of those programmes for landscape monitoring are listed in Table 2. In the second investigation, a multi-scale object-based monitoring and analysis of agricultural landscapes, involving parameters of various scales gives a wide possibility for data interpretation and analysis. The following three levels of agricultural landscapes were monitored: site level, measurement at a sample size of individual fields, focusing on the monitoring of ecosystem quality and on quality aspects of biodiversity relating to human pressure; farm level; and landscape level, focusing on functional and structural aspects of ecosystem dynamics in the larger context of adjacent environmental structure and process. In this investigation the relationships between the data sets of landscape elements and ecological parameters were tested.

3.2. Neighbourhood analysis

The methodology is based on the statistical description of point patterns and neighbourhood analysis (Upton and Fingleton, 1985, 1989; Haining, 2003). In this approach, the main tools are distance parameters or distance statistics. The density of sets is weighted according to their distance zones and neighbourhood. Monitoring sets are described by distance methods, assessing distances between points or the distance to geographical objects or factors. The pattern of monitoring is modelled on the basis of landscape districts, land cover, vegetation, or soil typology. MS Excel, MapInfo, Vertical Mapper, and CrimeStat (Levine, 2002) were used for analysis of the spatial model and data. The thematic maps were produced using MapInfo Professional. The basic data source was the national environmental monitoring programme.

One of the oldest distance statistics is the nearest neighbour index. It compares the distances between the nearest points with the distances that would be expected on the basis of chance (Ripley, 1981). The formula is simple to understand and to calculate. The distance to the nearest neighbour is calculated and averaged over all points.

$$d(\mathrm{NN})\frac{\sum_{i=1}^{N}((\min(d_{ij}))/N)}{(1/2)(\sqrt{A/N})} \quad (1)$$

where N is the number of points in the distribution, $\min(d_{ij})$ is the distance between each point and its nearest neighbour and A is the area of the region. If the observed average distance is the same as the mean random distance, then the ratio will be 1.0. If the index is greater than 1.0, there is evidence of dispersion. If the index is lower than 1.0 it shows clustering of the distribution. The nearest neighbour index is a measure of first-order spatial randomness. The K-order nearest neighbour indices are calculated and explored for the investigation of sets and for comparisons to be made, for example, between plant community and landscape monitoring.

The second function that was implemented is based on Ripley's K-function. This is an upper order nearest neighbourhood statistic, which provides a test of randomness for every distance from the smallest up to the size of the study area. Ripley's K-function is designed to measure second-order trends (Ripley, 1981). In fact, Ripley's K-function is the index of non-randomness. As a second-order statistic it shows how local clustering is opposed to a general pattern of the set over the region (O'Sullivan and Unwin, 2003). However, it is also subject to first-order effects, which means that it is not strictly a second-order measurement. Similarly to the nearest neighbour index, Ripley's K-function is applied in order to compare the monitoring sets.

Under unconstrained conditions, K is defined as:

$$K(d_s) = \frac{A}{N^2}\sum_i\sum_j I(d_{ij}) \tag{2}$$

$$d_s = \frac{R}{100} \tag{3}$$

where $I(d_{ij})$ is the number of other points, j, found within the distance d_s, added together over all points, i. R is the radius of a circle for the study area. $K(d_s)$ is transformed into a square root function. $L(d_s)$ is defined as:

$$L(d_s)\sqrt{\frac{K(d_s)}{\pi}} - d_s \tag{4}$$

3.3. Multi-scale object-based analysis for agricultural landscapes

The main objectives of the Estonian agricultural monitoring programme are to follow up and evaluate the environmental effects of land and agricultural reforms, to study changes in land cover types, especially fallow land and semi-natural areas, and to explain the connection between landscape structural indicators and the characteristics of ecological status of agricultural landscapes. The programme has a multi-disciplinary approach, has scales focusing on the spatial structure of the landscape, and includes aspects of biodiversity, cultural heritage, and human pressure on ecosystems.

Altogether nineteen study areas were strategically selected. The main criteria for the selection of study sites were:

- Distribution according to the Estonian landscape districts.
- Distribution throughout the country.
- Intensive and extensive areas as well as marginal areas of agriculture.
- Availability of complementary data.
- Relationship with other monitoring sites, especially with biodiversity monitoring networks.

Monitoring areas were selected in co-operation with the Ministry of the Environment and the Ministry of Agriculture. Monitoring commenced in 1996, and will cover the entire country on a 6-year rotation. By 2007 the test areas will have been recorded a second time and the actual change of agricultural landscapes can then be analysed. The test areas were mapped according to the classifications of areal, linear and point elements during the field studies. The size of the test areas to investigate land use, as well as the linear and point elements of landscape, was between 450 ha and 1200 ha, depending on the actual pattern of agricultural land-use. The main landscape elements, including arable lands, forested areas, pastures, grasslands, fallow lands, water bodies, parks and open pits were mapped on the land use plan or aerial photos on a scale of 1:10,000. Fallow lands were additionally described in terms of the time they had been fallow, and according to the dominant plant species. The results of field studies were digitised and encoded. Digitising and analyses were carried out according to the classification of areal, linear, and point elements using the software MapInfo. From these maps a number of indicators, such as Edge index (m/ha), length of linear elements per ha of monitoring area (agricultural land), number of point elements per ha of monitoring area (agricultural land), number of patches per ha, are calculated.

In the investigation of human pressure on agricultural land the following earthworm and soil microbial community parameters were selected and described: number of individuals and species of earthworms (*Lumbricidae*) per 1 m^2; maximum dominance in earthworm community (%); diversity of soil microbial and earthworm communities; total hydrolytical activity of soil micro-organisms; the number of colony-forming micro-organisms per 1 g of dry soil. Three survey sites were chosen in each monitoring area. To study the response of earthworm species to environmental factors, a linear ordination method, redundancy analysis (RDA) (ter Braak and Prentice, 1988) was used.

The flower visits of bumblebees were surveyed by using a standard quadrate-transect method (Banaszak, 1983; Teräs, 1985). Counts are carried out in all test areas, to be compared in pairs. Each locality included two transects (2 m × 1000 m)—one passing through a (semi-)natural habitat and the other through an agricultural habitat. Transects in semi-natural habitats passed through old (more than 20 years) late-successional annually mowed meadows, wooded meadows and forests. Agricultural transects passed through field boundaries, roadsides, pastures, orchards, clover, alfalfa, and oilseed rape fields. Both transects were divided into 20 m × 2 m plots.

4. Results and discussions

4.1. Distribution of sets for landscape monitoring

First, the categorical analysis is presented, which explores the distribution of monitoring networks on the basis of land cover, soils, and landscape districts. According to CORINE land cover (Meiner, 1999), 9% of all monitoring stations in Estonia are located in built-up areas, 39% in semi-natural areas, 43% in natural areas (excluding wetlands), 6% in wetlands and 3% in lakes and rivers. This distribution in general mirrors the distribution of land cover. The highest number of stations in a single cover type, 216, are situated in coniferous forests, 168 on land principally used for agriculture, and 129 stations in cultivated fields. In essence, the density is higher in built-up areas, where sampling strategy focuses on monitoring human impact (40 stations per 100 km^2). The representation of monitoring stations by land cover needs to be assessed by topical sets rather than as a whole (Table 3). For each stratified spatial realisation we need deeper

Table 3
Distribution of monitoring stations by land cover (CORINE land cover)

Land cover classes by CORINE	Monitoring network by environmental strata										
	Total	Stations per 100 km^2	Meteorology	Air	Groundwater	Rivers, lakes	Landscape	Plant community	Faunistic	Forest	Soil
Continuous urban	2	40.54		1	1						
Discontinous urban	76	17.27	9	5	47	5	3	3	4		
Industrial units	15	8.45	1		8	1		2	3		
Road and rail	1	2.88			1						
Port areas	3	30.75			3						
Mineral extraction	5	7.30			5						
Green urban	7	31.48			5	1	1				
Sport and leisure	1	5.80				1					
Non-irrigated arable	129	1.95	2	2	54	4	5	8	18		36
Fruit trees	1	4.90			1						
Pastures	58	1.99	2	1	27	2	2	10	6		8
Cultivation	48	2.93	3		33	1		3	4		4
Occupied by agriculture	168	4.95	7	11	95	22	7	9	16		1
Broad-leaved forest	64	1.45			16	3	2	16	18	9	
Coniferous forest	216	2.55		2	72	8	7	26	49	51	1
Mixed forest	107	1.27	2	2	38	3	6	5	23	27	1
Natural grassland	23	5.59			6	2	3	6	6		
Moors	7	4.46	1		2		1	3			
Beaches, dunes	6	9.28	1		2			2	1		
Forest-mineral	57	2.51	1	2	20	6	4	10	12	2	
Forest-swamp	31	2.21			6	1		8	15	1	
Marsh	9	2.80	2				1	3	3		
Fen	9	2.09						3	6		
Raised bog	20	2.08			3			7	9	1	
Water courses	13	39.27			9	3			1		
Water bodies	16	0.79			7	6			3		
Coastal lagoons	1	6.81					1				

ecological knowledge and additional data for evaluation. Faunistic monitoring is not widespread in wetlands. The monitoring set of plant groups intensively covers alvars in coastal lowlands. Natural grasslands are proportionally over-represented due to the targeted monitoring of rare and endangered species.

The forest monitoring set corresponds more or less to a proportional random selection throughout different forest types. 51 stations are situated in coniferous forest, 27 stations in mixed forest, and 9 in broad-leaved forest. Examining soil types, rendsic lepsol and skeletic regosol soil types are over-represented. On the other hand, stagnic luvisols and dystric histosols are under-represented. The main advantage of stratified assessment is that, even if the geographic data or some characteristics that the user is looking for are not available, knowledge from other areas and the identified patterns enable a compromise between needs and availability to be made. The exploration of areas having the same land cover or soil types enhances data mining techniques and best suits the available sampling set for our objectives. Classification of data by geographical attributes improves our ability to exploit common object- and field-based analysis functions.

The environmental monitoring networks in Estonia have not been established randomly, which a priori could guarantee that an event is located, surveyed and measured as a random sample. In some ways the topic-based summed-up monitoring may be considered incidental, because the sets of the sub-programmes are independent of each other. In assessing total density of networks, the monitoring stations are concentrated in the Tallinn area and in north-east Estonia, where human impact is intense, and to a lesser degree in Pärnu and west Saaremaa, which are covered by a dense biodiversity monitoring set. Large landscape districts are proportionally less represented in the total monitoring set, and small districts such as the lowland of the Gulf of Finland, Karula upland, and Palumaa are more intensively surveyed. According to geographical distribution, coastal lowlands have the most intensive coverage in monitoring sets (Fig. 2).

4.2. Neighbourhood indices

The examples presented in the previous section demonstrate that categorical analysis is informative, having potential value for decision-making. Nine topical sets were assessed through nearest neighbourhood indices (Fig. 3). As Estonia's landscape is composed of a variety of landscape types and the landscape is in flux, different distances and neighbourhood relations will have to be present in the monitoring network. In general, pollution related sets tend to cluster around 'hot spots', with few reference areas being represented. The groundwater monitoring set is the most closely clustered, with a nearest neighbourhood index of 0.18 (the average distance to the nearest station is 1.2 km). The set forms clusters in north-eastern Estonia, in Pandivere, a nitrate sensitive area, and in the Tallinn metropolitan district, an area with a significant human impact. The monitoring sets of plant and animal species (flora and fauna) are clustered in protected areas. Compared with other sets, the meteorological

Fig. 2. Distribution of landscape monitoring set and density of all monitoring sets by landscape regions.

monitoring sets are the most dispersed, and, according to the Euclidean measures, even over-dispersed (the average distance to the nearest station is 33.3 km; the random nearest neighbour distance is 25.8 km). Landscape monitoring also shows higher dispersion. As an exception, a geometrically regular monitoring set is implemented in the International Co-operative Programme (ICP) forest monitoring programme, which is set up across Europe on a grid of 16 km × 16 km. Estonia has 90 monitoring stations with 2136 observation trees. Consequently, regular, dispersed, aggregated, and random patterns are observed in the Estonian landscape monitoring set.

Explaining the curve of *K*-order nearest neighbour indices (Fig. 4), the groundwater monitoring set is clustered up to and including the fourth rank. Clusters of air monitoring are relatively dispersed and located in different parts of Estonia. Paired sites affect the distribution of landscape and inland water monitoring sets. In general, after the fourth rank nearest neighbour, the differences between sets become less pronounced.

It is possible to compare the distribution of *L* between the sets and for various baseline landscape characteristics. The Ripley's *K*-function describes the hierarchy of clustering (Fig. 4). Clustering is expressed clearly in the groundwater monitoring set, having a radius of 30 km for groundwater bodies. Hierarchical clusters are clearly described in the plant species monitoring set, where the density of the point pattern increases up to a search radius of 25 km, which represents the size of larger protected areas. The 80 km buffer expresses the distance between nature protection areas. For smaller sets, like those for meteorology and soils, the curve shows an increase in clustering over long distances. According to Ripley's function, monitoring of fauna and forest sets are random and dispersed over longer distances.

Designing monitoring networks to be spatially more efficient is one of the keys to upgrading monitoring methods and decision-support systems. The issue is not just to establish new monitoring stations, but to relocate stations towards unmonitored areas. Fig. 5 shows the total density map of monitoring sets of different monitoring themes, at a 50 km search radius. As a density map has a linear dependence on the width of the search buffer, the methodology of the monitoring and the spatial func-

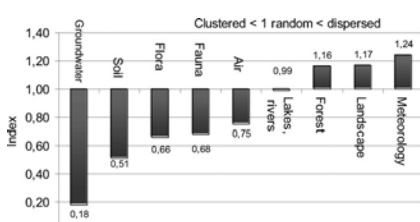

Fig. 3. Nearest neighbour index of the Estonian environmental monitoring set.

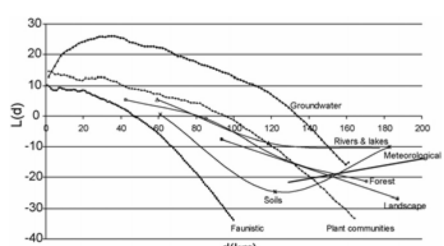

Fig. 4. Density of monitoring stations according to Ripley's *K*-function.

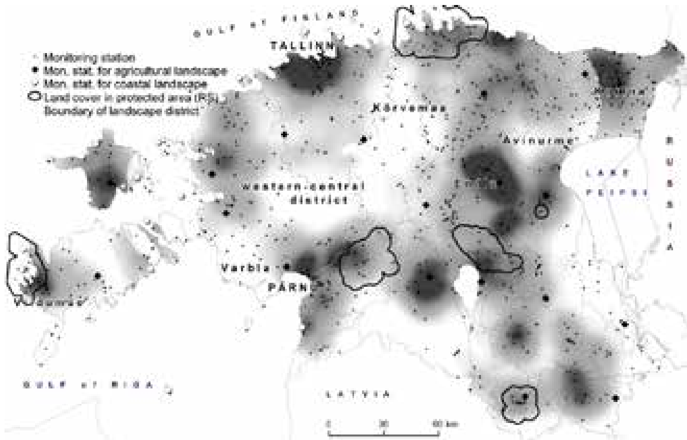

Fig. 5. Total density of monitoring sets (50 km search radius, dark:high density, light:low density).

tion of the environmental phenomenon should be considered. Regarding the 50 km search radius, it was assumed that a transfer function could be applied for such a distance. Also, 50 km could be taken as the average maximum distance between the monitoring stations. According to the model, stratified environmental information is provided for landscapes in the metropolitan areas near Tallinn, Pärnu and in north-eastern Estonia (Kurtna Lakes). Also, the Endla and Viidumäe national parks are certainly covered. Areas that are more sparsely and less certainly covered by monitoring information are the western-central part of Estonia and the border areas with Latvia. Smaller "uncovered" areas are found in northern Kõrvemaa, in Avinurme and around Varbla.

4.3. Assessment of applications in agricultural landscape monitoring

The first results of the Estonian agricultural landscape monitoring programme can be considered successful, in that it has achieved its initial aims of reporting according to the selected parameters and indicators on landscape structure and biodiversity. Monitoring of agricultural landscapes is supported by datasets of environmental monitoring. The neighbourhood analysis provides a modelling technique and statistical module for obtaining parameters for comprehensive landscape analysis (Fig. 6). The chosen multi-scale object-based methods provide a good overview of the level of human pressure on different categories of agricultural land and for defining priorities for landscape management. For example, it is stated among the results of the monitoring that the species composition and abundance of bumblebees was, to a great degree, determined by landscape structure (Sepp et al., 2004). The main gradient in bumblebee species distribution is connected with naturalness of the monitoring areas. The number of bumblebee species and abundance in agricultural habitats was smaller than in (semi-)natural habitats. The most important species of bumblebees in grouping study sites into semi-natural or anthropogenic ones are *Bombus pratorum*, *B. sylvarum*, *B. lapidarius* and *B. veteranus*. The method based on the assessment of the numerical composition of bumblebee species describes the human impact on the landscape scale adequately. The most important landscape features correlating with the distribution of bumblebee species are the length of ecotones between agricultural land and mixed forests, mixed forests, and wetlands, on the one hand, and the length of ecotones between agricultural land and broad-leaved forests, cultivated grasslands, and legumes. On the basis of soil microorganism and earthworm data the different types of agricultural land (arable land, fallow land, cultivated grassland, natural grassland) are well described. Lands that were abandoned 3–4 years ago are still in depression—the number of earthworm individuals is relatively low and the number of earthworm species is three to five (Sepp et al., 2005).

The strategic approach in the selection of monitoring areas based on landscape districts is cost-effective but it has its own limits concerning the interpretation of the results. It seems that the set of agricultural monitoring areas may not be sufficient to summarize monitoring results per landscape district. Either we should increase the area of monitoring sites or increase their number. At the same, the methods chosen for data collection have proven efficient and, on the basis of measured parameters, we can evaluate landscape change and human pressure on landscape structure and biodiversity. Complementary data on landscape components could be obtained from other environmental monitoring programmes directly or by applying different methods of extrapolation, like the neighbourhood method, using the spatial unit of landscape district. The neighbourhood method could be

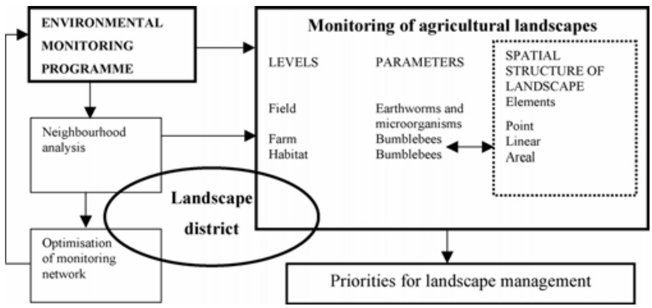

Fig. 6. Framework for applied integrated landscape monitoring.

applied for the optimisation of monitoring sets discussed in the next section.

4.4. Designing the monitoring network

In recent years and definitely in the near future the monitoring network will be enlarged due to the European Union directives and networks (Folving et al., 2001; Bastian et al., 2002; Groom, 2004). It is arguable whether the data of the national monitoring network is sufficient and cohesive enough for calculating any national averages, although these are generally used as indicators, or indexes in European reports. This may affect the design of the monitoring network. When optimising the monitoring network, different models have been applied, which do not only deal with the spatial features of the monitoring, but also with the complexity of the subject.

In optimising the monitoring set, different location-based models have been applied. The discussion on methods and approaches continues in searching for a key. Do we assume representation by the typological classes, or do we fill the pattern in 'hot spots', or do we seek to achieve total national coverage? When studying these spatial relations, the primary factor is the phenomenon of interest itself, and the complexity of the landscape makes this a non-trivial problem. The criteria of selection for monitoring methods and sampling strategy must adequately follow spatial relationships for the subject as well as for wider purposes (Fuller et al., 1993; Fjellstad et al., 2001; Dramstad et al., 2002; Lausch and Herzog, 2002). Therefore, upgraded monitoring methods, spatial analysis methods, and behaviour and spatial functions of the phenomenon are applied for multiple purposes. Also, remote sensing could assist in enabling an integrated analysis in applied environmental studies (Ihse and Blom, 1999). In the case of covering the whole of Estonia, but also in the case of small test areas, the ground-level monitoring network can be connected with distance monitoring, which together enables an integrated analysis in applied environmental studies.

Human impacts on the agricultural landscape often occur on a site-specific basis. If we try to mitigate environmental impacts on a site-specific basis, it is difficult to account for the cumulative effects that result (Brandt et al., 2002; Sepp et al., 2004). Some species are favoured by a large number of forest or field edges, others by homogeneous landscapes (Forman, 1995; Bender et al., 2003). Some landscapes are characterised by high heterogeneity and others by low heterogeneity (at a specified scale of measurement). Again, the value of spatial heterogeneity as a monitoring measure resides in the fact that it can indicate landscape change. How to respond to the information or to set targets will be value judgements that must be made for the area in question.

A particular problem for environmental statistics is the spatial unit to which they refer. Whereas socio-economic indicators are usually available for administrative entities or areas, many environmental phenomena often manifest themselves regardless of administrative boundaries (Brandt et al., 1994; Dramstad et al., 2002). Relating environmental indices to districts delimited according to ecological criteria (landscape districts, catchments, landscape types, etc.) would increase their sensitivity and interpretability. Socio-economic indicators must be made available at the level of landscape districts, and administrative structures requested for the implementation of measures must also be created at this level. These structures must then coordinate their actions with the existing administrative bodies. Whether they are related to eco-regions or administrative units, landscape metrics need to be harmonised (Lausch and Herzog, 2002).

5. Conclusion

When establishing a system for landscape monitoring, it is essential that the landscape definition is suited to the phenomenon and process under consideration and that regional context is taken into account. In practise, landscape-monitoring programmes have different objectives, and the concept of 'landscape' used in monitoring also varies widely. With respect to multiple targets and methodologies, data for landscape analysis could and should be derived not only from special landscape monitoring programmes but also from other environmental monitoring sets, such as biodiversity, forest, soil, water, and integrated monitoring. A key benefit from the use of the latter is that they are legacy sets of intended surveys, produced with a specific purpose. Altogether, there are 11 sets of monitoring sub-programmes with approximately 1300 stations in Estonia.

This article assesses the neighbourhood of the Estonian monitoring network as a whole, in order to test the availability of characteristics of the landscape from multiple sources. The analysis is constructed so that the distance and proximity methods related to topical sets are synthesised for the sampling set of landscapes. The spatial analysis associated with landscape types and districts on the national level follows neighbourhood methods. Targeted supplementary analysis by in-depth methodologies of landscape monitoring makes available a full package of data on landscape domain. The combined use of the stratified topical approach of environmental monitoring and of landscape metrics embedded in understandings of spatial pattern can be used to support the monitoring of landscapes.

The Estonian agricultural monitoring programme can be considered successful and justified for its purpose. The chosen methods provide a good overview of the level of human pressure on different categories of agricultural land. Based on the experience gained from the implementation of the monitoring method, the following parameters have been chosen for characterising the human impact on agricultural landscapes: first, at a field level, individuals and species of earthworms (*Lumbricidae*) per 1 m^2, diversity of soil microbial and earthworm communities, total hydrolytical activity of soil micro-organisms and the number of colony forming micro-organisms per 1 g of dry soil; at a district level, the numerical composition of bumblebee species is the most informative parameter. Selected and mapped landscape features, agricultural and non-agricultural land cover categories, number and length of different linear elements, etc. clearly distinguish anthropogenic areas from semi-natural areas. A more thorough evaluation of the extent to which the monitoring programme has fulfilled its objectives cannot be made until the second cycle of national inventory has been completed. A multi-scale object-based monitoring and analysis of landscape gives a good overview of human pressure and landscape change. In the next cycle we should increase the number of monitoring sites, and socio-economic indicators must also be include at the level of landscape districts or administrative units.

The adequacy of landscape monitoring according to the spatial relation of the environmental monitoring set is explored by landscape district. In our prototype model of neighbourhood analysis, regarding the scope and objectives of the programme, various sampling approaches are set up to survey qualitative landscape parameters. Categorisation of data by geographical attributes improves our ability to exploit common object- and field-based analysis functions. The method used enables us to make decisions by identifying and interactively packaging comprehensive data structures on the level of landscape district. Further, data mining techniques can be enhanced according to land cover type, soil type or water basin. The validity and transferability of the method to match different data sources at different sites is discussed. In Estonia's case, a regular monitoring grid is only available for forest monitoring. All other sets are based on their own monitoring methodology, some of which aim to achieve overall national geographical coverage, some of them, to test different landscape districts. The monitoring sets aiming to acquire data on human impacts are clustered in metropolitan areas. Water monitoring sets are clustered around river basin areas. Biodiversity sets can easily be applied as data sources for landscape monitoring in national parks and protected areas. The representation on landcapes and land cover types is rather different. For that reason the application of the transfer functions needs further investigation and modelling on a small and meso-scale level. Small sets having less than 50 stations are biased, and tests have not found these data to be statistically significant.

A systematic approach focused on landscapes helps us to optimise the monitoring sets as a whole in order to achieve a coherent and efficient layout of monitoring sets for Estonia. For example, the biodiversity set needs further expansion in the southern uplands. Surface-water monitoring requires a more extensive set in western Estonia and in Saaremaa. A strategic approach for selecting monitoring stations is statistically preferable, because proportional samples of districts, which are relatively smaller for large homogeneous districts, are used. Also the procedure is more cost-effective, because large uniform areas require less sampling. An important addition to this work could be the linking of the geo-referenced monitoring data to the Estonian square kilometres database.

Critical issues that remain are the categorisation and choice of appropriate spatial units that will allow for an integration of landscape indicators that could potentially relate to cross-border phenomena and socio-economic indicators that are usually available for administrative entities or areas. The selection of a manageable set of indicators that embraces the structural properties of landscapes is another requirement for the successful integration of different sets. Also, standardised and harmonised data processing techniques are vital for the spatial and temporal comparability of results.

The potential of integrated methods for landscape monitoring should be further examined in relation to neighbourhood analysis. Applicability of modern automated techniques, which are initiated by management needs, depends on conceptual maturity and flexibility in data management.

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25.10.08 – 24.01.12	Baltic Green Belt, Baltic Sea Region Programme 2007–2013, 8-2/T9018PKPK
01.04.08 – 31.03.12	EBONE - European Biodiversity Observation Network: a project to design and test a biodiversity observing system integrated in space and time, 8-2/T8043PKPK
11.05.11 – 01.04.12	Conservation Management Plan for Haanja Nature Park (2012–2021), 8-2/T11059PKMH
01.05.10 – 30.09.10	Collection of data about indicators "Changes in landscape structure in point-, line- and polygon-elements", 8-2/10070PKPK
01.01.09 – 31.03.10	Development of Conservation Management Plans according to the principles of Natura 2000 and EU Landscape Convention, 8-2/T9009PKPK
01.11.09 – 15.04.11	Development of methods for assessing the impact of visitation load to protected objects and improvement of decision-support systems, 8-2/T9119PKPK
01.01.04 – 31.12.07	Relationship between numbers and distribution of waders and soil biota in coastal and floodplain meadows, ETF6005

26.05.09 – 12.11.09	Implementation of the sub-program of the national wildlife diversity and landscape monitoring program "Monitoring of agricultural landscapes", 8-2/T9070PKPK
01.07.10 – 11.11.10	Implementation of the sub-program of the national wildlife diversity and landscape monitoring program "Monitoring of agricultural landscapes", 8-2/10110PKPK
01.07.11 – 10.11.11	Implementation of the sub-program of the national wildlife diversity and landscape monitoring program "Monitoring of agricultural landscapes", 8-2/T11085PKMH
17.12.10 – 15.05.11	Conservation Management Plan for Võlumäe-Linnamäe landscape protected area, 8-2/T10217PKPK

Presentations at international conferences

14 – 16.03.2011	International Conference on Research and Management of the Historical Agricultural Landscape, Viničné, Slovak Republic: II. Plenary session: Historical agricultural landscape – source of landscape diversity and biodiversity, 15 March 2011. Oral presentation: Transformation of rural landscapes in Hiiumaa since 1956: consequences to open and half-open semi-natural habitats. III. Plenary session: Historical agricultural landscape in globalisation period – development trends and opportunities, perception, legislative, economic and political tools of preservation and sustainable utilisation, 16 March 2011. Oral presentation: Open and half-open semi-natural habitats in Hiiumaa: development trends and legislative, economic and political tools for sustainable use and preservation.
03 – 06.09.2010	International Conference on Landscape structure, functions and management: response to global change organised by CZ-IALE, Brno, Czech Republic. Poster presentation (co-author):

	EU impact on land use dynamics in Estonia. International Conference in Landscape Ecology.
16 – 18.06.2008	The Third International Conference on Evolution, Monitoring, Simulation, Management and Remediation of the Geological Environment and Landscape, New Forest, UK. Oral presentation: Land use changes on Hiiumaa Island (Northwestern Estonia) in the last fifty years.

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Teaduskraad

Keskkonnateaduse ja –poliitika magister, Manchesteri Ülikool ja Kesk-Euroopa Ülikool, 2000

Teadustöö põhisuunad

Maastikuökoloogia; maastikumuutus ja elustik; ökoloogiline võrgustik; bioindikatsioon

Osalemine uurimusprojektides

01.01.07 – 31.12.12	Antropogeensed mõjud biotoopidele ja maastikele: biootilised ja abiootilised markerid, sihtfinantseeritav projekt: SF1090050s07
25.10.08 – 24.01.12	Balti rohevöö, Baltic Sea Region Programme 2007–2013, 8-2/T9018PKPK
01.04.08 – 31.03.12	EBONE - European Biodiversity Observation Network: a project to design and test a biodiversity observing system integrated in space and time, 8-2/T8043PKPK
11.05.11 – 01.04.12	Haanja looduspargi kaitsekorralduskava koostamine aastateks 2012–2021, 8-2/T11059PKMH
01.05.10 – 30.09.10	Indikaatori "Maastiku struktuuri muutused punkt-, joon- ja pindelementides" andmete kogumine, 8-2/10070PKPK
01.01.09 – 31.03.10	Maastike kaitsekorraldusmudeli arendamine vastavalt Natura 2000 elupaikade kaitsele ja EÜ maastike konventsioonile - I etapp, 8-2/T9009PKPK
01.11.09 – 15.04.11	Metoodika arendamine kaitsealuste objektide külastuskoormuse mõjude hindamiseks ning kaitsekorralduslike otsuste tegemiseks, 8-2/T9119PKPK
01.01.04 – 31.12.07	Ranna- ja luhaniitude kurvitsaliste arvukuse ja leviku seos mullaelustiku komponentidega, ETF6005
26.05.09 – 12.11.09	Riikliku elusloodusе mitmekesisuse ja maastike seire alaprogrammi "Põllumajandusmaastike seire" täitmine, 8-2/T9070PKPK
01.07.10 – 11.11.10	Riikliku elusloodusе mitmekesisuse ja maastike seire alaprogrammi "Põllumajandusmaastike seire" täitmine 2010, 8-2/10110PKPK
01.07.11 – 10.11.11	Riikliku elusloodusе mitmekesisuse ja maastike seire allprogrammi seiretöö "Põllumajandusmaastike seire" täitmine, 8-2/T11085PKMH

17.12.10 – 15.05.11 Võlumäe-Linnamäe maastikukaitseala kaitsekorralduskava koostamine, 8-2/T10217PKPK

Ettekanded rahvusvahelistel konverentsidel

14 – 16.03.2011 Rahvusvaheline konverents: Research and Management of the Historical Agricultural Landscape, Viničné, Slovak Republic:
II Plenaaristung: Historical agricultural landscape – source of landscape diversity and biodiversity, 15 March 2011. Suuline ettekanne: Transformation of rural landscapes in Hiiumaa since 1956: consequences to open and half-open semi-natural habitats.
III Plenaaristung: Historical agricultural landscape in globalisation period – development trends and opportunities, perception, legislative, economic and political tools of preservation and sustainable utilisation, 16 March 2011. Suuline ettekanne: Open and half-open semi-natural habitats in Hiiumaa: development trends and legislative, economic and political tools for sustainable use and preservation.

03 – 06.09.2010 Rahvusvaheline konverents: Landscape structure, functions and management: response to global change, organised by CZ-IALE, Brno, Czech Republic. Poster ettekanne (kaasautor): EU impact on land use dynamics in Estonia. International Conference in Landscape Ecology.

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LIST OF PUBLICATIONS

1.1. - Scholarly articles indexed in the ISI Web of Science database

Leito, Aivar; Truu, Jaak; Õunsaar, Maris; Sepp, Kalev; **Kaasik, Are**; Ojaste, Ivar; Mägi, Eve 2008. The impact of agriculture on autumn staging Eurasian Cranes (*Grus grus*) in Estonia. Agricultural and Food Science 17(1), 53–62.

Roose, A.; Sepp, K.; Saluveer, E.; **Kaasik, A.**; Oja, T. 2007. Neighbourhood-defined approaches for integrating and designing landscape monitoring in Estonia. Landscape and Urban Planning 79, 177–189.

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1.2. - Peer-reviewed articles in other international research journals

Kaasik, Are; Sepp, Kalev; Raet, Janar; Kuusemets, Valdo 2011. Transformation of rural landscape in Hiiumaa since 1956: consequences to open and half-open semi-natural habitats. Ekológia (Bratislava) 30(2), 257–268.

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3.1. - Articles/chapters in books

Kaasik, A.; Raet, J.; Sepp, K.; Leito A. & Kuusemets, V. 2008. Land use changes on Hiiumaa Island (North-western Estonia) in the last fifty years. In: Ü. Mander, C.A. Brebbia, J.F. Martin-Duque (Eds), WIT Transactions on the Built Environment, Vol. 100, Geo-Environment and Landscape Evolution III: Third International Conference on Evolution, Monitoring, Simulation, Management and Remediation of the Geological Environment and Landscape, 16–18 June 2008, The New Forest, UK. WIT Press, Southampton, pp. 173–182.

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3.4. - Articles/presentations published in conference proceedings

Sepp, K.; Jagomägi, J.; **Kaasik, A.**; Nikodemus, O.; Kreilis, M.; Strazdina, B.; Busa, V.; Vaiciunaite, R.; Mierauskas, P.; Gulbinas, Z.; Ramonas, A.; Kurlavicius, P.; Gudžinskas, Z.; Rašomavicius, P.; Ivinskis, P.; Kesminas, V.; Mickevicius, E. 2002. Development of National Ecological Networks in the Baltic Countries in the framework of the Pan-European Ecological Network. IUCN Office for Central Europe, 165 p.

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LAST FIVE DEFENCES

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MICROSITES AND TREE REGENERATION DYNAMICS:
PROLONGED STORM EFFECTS IN HEMIBOREAL MIXED FOREST
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