DISSERTATIONES GEOGRAPHICAE UNIVERSITATIS TARTUENSIS

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OPTIMISATION OF ENVIRONMENTAL MONITORING NETWORK BY INTEGRATED MODELLING STRATEGY WITH GEOGRAPHIC INFORMATION SYSTEM — AN ESTONIAN CASE

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The Faculty Council of Biology and Geography, University of Tartu, has on the April 28, 2005 accepted this dissertation to be defended for the degree of Doctor of Philosophy (in Geography).

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The thesis will be defended at the University of Tartu, Estonia, on August 22, 2005 at 10.15 in the Scientific Council room in university main building, Ülikooli 18.

The publication of this dissertation has been funded by the Institute of Geography, University of Tartu.

ISBN 9949–11–108–0 (trükis)
ISBN 9949–11–109–9 (PDF)

Tartu Ülikooli Kirjastus
www.tyk.ee
Tellimus nr. 264

CONTENTS

ABSTRACT

The dissertation proposes a modelling strategy using a geographical information systems approach for analysing the adequacy of environmental monitoring sets, by spatially relating monitoring stations and proposing optimisation methodology for environmental monitoring of air pollution, landscapes and hazardous substances. The aim is to describe the point patterns of environmental monitoring networks in Estonia, assess their neighbourhood, configuration and representation according to certain criteria, and to analyse the coherence of the environmental strata and monitoring sets for their efficient allocation. The methodology is based on dispersion modelling, on the statistical description of point patterns, on neighbourhood analysis and on distribution analysis. A prototype system based on the data model is implemented in Mapinfo Professional®, Vertical Mapper, Idrisi and CrimeStat. The prototype is illustrated and tested by three case studies: air pollution, landscape monitoring and monitoring of hazardous substances. All three case studies are based on Estonia's environmental monitoring system and its spatial presentation. The case study on air pollution in north-eastern Estonia establishes cause-effect links between production, emissions, and ambient air quality from a single-plant case to complex situations. The study explores the compounding effect of air pollution and spatial sensitivity to changes, building predictive models of SO_2 emissions. The spatial analytical capabilities of a raster system using approximate diffusion models for SO_2 pollution load in a 1 km grid are explored. The series of images are examined as a whole and changes are explained according their significance and power. Changing industrial structure may result in completely different types of air pollution maps and a different type of behaviour pattern in pollution load images. A spatially-explicit method of network design for landscape monitoring and sampling strategies combines stratified and multi-scale agricultural landscape monitoring, testing the hypothesis of the importance of neighbourhood. The nearest neighbour index and Ripley's K-function are applied. The proximity to certain land uses, land cover, soils, vegetation and human impacts is investigated to assess how landscape features are covered by different strata of monitoring data and how the current pattern of monitoring network represents the landscape features. It is concluded that proportional stratified sampling requires fewer sites for large homogenous inland landscape regions. A systematic approach, focused on landscape classes, helps to integrate the monitoring set as a whole and to achieve a coherent layout of monitoring networks for landscape research. As another case study, distribution of hazardous substance is presented. Monitoring of hazardous substances focuses on problem areas reflecting intensive human impact. During the last decade the status of environment-related priority hazardous substances has continuously improved in Estonia. According to monitoring results, the concentration of hazardous substances in sediments and in surface water remains low in the majority of

Estonian rivers, and their status by European standards is classified as high and good. Using bio-accumulation methodology, the concentrations of hazardous substances found in Baltic fish in the Estonian coastal sea remain below standards established by the FAO/WHO for food. Specifically, the patterns of polychlorinated biphenyls (PCB) in the grey seals (*Halichoerus grypus*) from the Baltic, North-East and Eastern England, and the St. Lawrence Estuary (Canada) are examined. The patterns differ between juveniles and adult animals, but the gender of adults and geography do not appear to play a role. The key to the improvement of monitoring is the integration of source-oriented and load-oriented approaches, since both are lacking full-scale consistent data coverage. Spatial factors may have implications for the design of monitoring networks assuming national coverage. Evaluation of monitoring sites that do not match methodological or least-distance criteria can provide suggestions for the optimisation of the network.

Keywords: environmental monitoring; environmental modelling; dispersion modelling; distance statistics; nearest neighbourhood; spatial pattern; spatial variation; sampling; air pollution; landscape monitoring; hazardous substances; Estonia; the Baltic Sea.

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LIST OF PAPERS

This dissertation consists of a comprehensive summary that is based on following papers I–V, which are referred to in the text by their Roman numerals:

I **Roose, A.**: **Spatial Analysis of Industrial Impacts on Air Pollution: an Estonian Case.** In: Brebbia, C. A. & Martin-Duq, J.F. (eds.). Air Pollution 2002 Proceedings, 1–3 July 2002, Segovia, WIT Press, pp. 43–52.

II **Roose, A.**: **Optimal set for monitoring the environment in Estonia — neighbourhood analysis.** In: Latini, G., Passerini, G. & Brebbia, C.A. (eds.). Development and Application of Computer Techniques to Environmental Studies, Envirosoft X, WIT Press, 2004, pp. 13–22.

III **Roose, A.**, Sepp, K., Saluveer, E., Oja, T.: **Neighbourhood-defined approaches for integrating and designing landscape monitoring in Estonia.** Landscape and Urban Planning — submitted 09.08.2004.

IV **Roose, A.**, Roots, O.: **Monitoring of priority hazardous substances in Estonian water bodies and in the coastal Baltic Sea.** Boreal Environment Research 10, 89–102. 2005.

V Roots, O., Zitko, V., **Roose, A.**: **Persistent organic pollutant patterns in grey seals (*Halichoerus grypus*)**. Chemosphere — accepted 14.01.2005, article in press, available online 8 March 2005.

Author's contribution

I The author is fully responsible for the theoretical part, data collection, modelling and analysis and for writing the manuscript.

II The author is fully responsible for the theoretical part, data collection, modelling and analysis and for writing the manuscript.

III The author initiated the article, set up the theoretical framework, performed data collection, modelling and analysis related to the neighbourhood approach. The author prepared the first draft of the paper.

IV The paper was planned jointly by the author and O.Roots. The author is fully responsible for the structure and synthesis of data in the paper. The author contributed following parts of the article: introduction, study area, geographical distribution of hazardous substances, and survey on heavy metals.

V The paper was initiated by O.Roots and V.Zitko. A.Roose assisted in geographical analysis, writing overview on spatial distribution and producing maps for the manuscript.

Interactions between the papers quoted

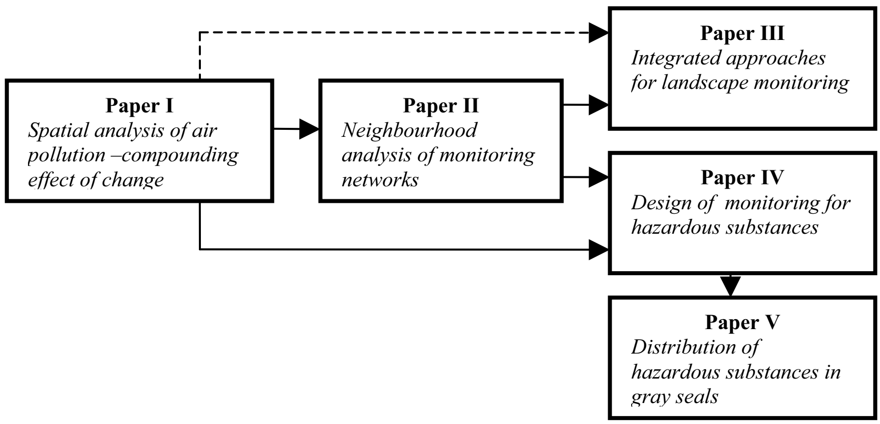

Figure 1. The structure of articles in the thesis

Below, content and comments on the background of the papers are given in logical order:

I Spatial Analysis of Industrial Impacts on Air Pollution: an Estonian Case.

An initial case study on spatial modelling of air pollution was carried out to explore several interlinked sub-models between production, processing, emissions, and ambient air quality. The modelling exercise was performed on meso-scale dispersion of pollutants for North-East Estonia. The introductory stage of research investigated empirical methods of building predictive models of SO_2 emission changes for the selected area. The spatial analytical capabilities of a raster system based on Idrisi software, using approximate diffusion models for SO_2 pollution load in a 1 km grid were tested. The series of images were examined as a whole and changes were explained according their significance, power and sensitivity. Surrogate indicators proposed in the final part demonstrated spatial preferences for air pollution control policies.

II Optimal set for monitoring the environment in Estonia — neighbourhood analysis.

The paper launched a new modelling approach for analysing the spatial features of environmental monitoring sets. The concept of neighbourhood, including notions to distance, adjacency and interaction, is employed. Point pattern of environmental monitoring is decribed using various measures and statistics, specifically the nearest neighbour index and Ripley's K-function. A prototype system was developed using Mapinfo Professional®, Vertical Mapper and CrimeStat, also MS Office™. The prototype is illustrated and tested by a case study based on Estonia's environmental monitoring system. The point patterns of the environmental monitoring network in Estonia are characterised. The spatial optimisation of the network was discussed. The modelling was done with the intention of avoiding 'real' environmental variables, in order to verify that the neighbourhood statistics, the same methods for all monitoring networks, can explore the key for spatial relationships. The designed monitoring network is supported by a neighbourhood approach.

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

Based on the theoretical and modelling experiences described in **Paper II**, this paper compares conventional and neighbourhood-defined approaches for a sound landscape monitoring programme. Criteria for selecting study areas, hierarchical levels, and techniques of data collection and analysis were identified in the framework of a case study of landscape monitoring in Estonia. Two major methodologies, neighbourhood analysis and multi-scale object-based methodology of agricultural landscapes were assessed. Significant findings show how the current pattern of the monitoring network represents the landscape features. Tests carried out in local and regional pilot areas help to make a monitoring methodology robust before it is applied at a national level. Both complementary methods are directed at network design for monitoring mosaic landscapes. A systematic approach of categorical mapping focused on landscape classes, land cover, and soils. In this way, the paper advances several model tests applied initially in **paper II**. 3D models of the monitoring network were tested in UTM grid and Estonian square kilometres databases.

IV Monitoring of priority hazardous substances in Estonian water bodies and in the coastal Baltic Sea.

The paper introduces the distribution of priority hazardous substances, persistent organic pollutants and heavy metals in Estonian water bodies and in the coastal Baltic Sea. This work began with the collection and evaluation of monitoring data on priority hazardous substances in Estonia. The spatial distribution was described in terms of relationships between spatial entities. The paper focused on problem areas in metropolitan areas of Tallinn and in the oil-shale region of North-Eastern Estonia in order to reflect intensive human impact. Several mapping techniques were employed to delineate the distribution of hazardous substances and to outline monitoring clusters. Similarly to **Paper I**, the paper discusses the relationship between emissions and impacts as they are reflected in the monitoring network. The specific study area is also the oil-shale region. The article proposes further steps to improve monitoring.

V Persistent organic pollutant patterns in grey seals (*Halichoerus grypus*).

The paper develops the ideas and findings described in **Paper IV** concerning the monitoring of hazardous substances, and specifically examines the patterns of polychlorinated biphenyls (PCBs) in grey seals (*Halichoerus grypus*) in an inter-regional comparison of the Baltic, North-East and Eastern England, and the St. Lawrence Estuary in Canada. The profiles of polychlorinated biphenyls were examined by Principal Component Analysis. The patterns of geographical distribution of juveniles and adults were examined.

CONTENTS OF COMPREHENSIVE SUMMARY

A doctoral dissertation comprises a summary of a number of papers. The following is the comprehensive summary of the quoted papers.

1. INTRODUCTION

The introduction outlines the framework for the system developed in the dissertation, and the problems, scope and potential role of the system, prior to fuller discussion in subsequent chapters.

Geographic Information System for environmental modelling

Successful environmental policies need to be underpinned by relevant and reliable information. There is often a gap, however, between the information available, methodologies for assessment, and end-user information for sound policymaking (Longley *et al.*, 1999; Skidmore, 2002). As the practice of environmental monitoring is developing, it is becoming more standardised. The quality of environmental monitoring, the level of sophistication and technical know-how is increasing, although the overall quality of data assessment available to policymakers is still far from what would be desirable (Parr *et al.*, 2003). An efficient allocation of resources requires a consistent concept of the handling and integration of data at defined working scales that are compatible with the operational levels of environmental planning and management (Clarke *et al.*, 2002).

Expert systems, computer systems crystallising the way an expert solves the problems, have demonstrated considerable appeal in many research areas (Rodriguez-Bachiller and Glasson, 2004). One of them, the geographic information system (GIS), specified here, is becoming more widespread in local and central government agencies as well in private companies, but it is sometimes not very clear how to make the investment into GIS pay off, how to run it professionally, efficiently, in a coherent manner regarding the stated environmental problem. Needless to say, valuable information is still overwhelmingly extracted from time-series rather than detailed maps.

Against this background, the proposition behind the work presented here is that these two areas: environmental monitoring and geographical information systems, are potentially complementary and there would be mutual benefits if they could be brought together. The sources of data and expertise to solve the problem should be available and accessible. This work suggests new integrated approaches for effective policy implementation through a comprehensive monitoring system, plunging directly into application work, with the production of a prototype.

The main purpose of this research has been to point out the complementarity of these areas of environmental monitoring, and the arguments can be summarised as follows:

- GIS practice is growing at a fast pace, and many of the actors involved are finding it difficult to cope (Skidmore, 2002).

- The quality of environmental monitoring is still far from satisfactory. One reason for lower quality is still relative ignorance about sampling strategies and about the spatial aspects of monitoring (Wiersma and Wiersma, 2004). However, interest in spatial applications is widespread and it is timely to have a closer look at this possibility.

- GIS is becoming more efficient and analytical capabilities are continually improving. However, GIS are still to some extent prisoners of their cartographic past, so that part of their functionality was initially directed towards solving cartographic problems, and their analytical capability was somewhat neglected (Maguire and Dangermond, 1991; Longley *et al.*, 2005).

Data availability and GIS applicability

The dissertation acknowledges problems in all three key aspects that have marked different stages of GIS development:

1. Research issues in solving specific mapping and database problems
2. Expertise — the lack of sufficient numbers of professionals to use and apply GIS
3. Data: this refers more to data quality, data availability, and cost

Two benchmark publications (Maguire *et al.*, 1991 and Longley *et al.*, 2005) summarise most of the research and development issues. Relational databases are becoming more familiar. GIS take the idea one step further by making it possible to include spatial positioning as one parameter in the database (Maguire, 1991). GIS can most simply be seen as spatially referenced databases.

Any spatial dataset, including monitoring data, provides an abstraction of a complex reality. The principal question is that the analyst needs to be alert not only to the problem but also to how it may impact differentially across the study area or between study areas. It is critical to achieve fitness for purpose.

As usual, environmental management requires three different kinds of information. First, the status of environmental indicators or variables needs to be estimated — *what is where*. The second type of information is change over time — *what is changing where*. Third, one is interested in forecasting, both in spatial as well as temporal resolution — *what will be where*. The last question can not be answered with great precision. Also, space-temporal modelling is very often limited due to lack of data. The lack of consistent and up-to-date information on the state of the environment has been identified as a one of constraints for the implementation of environmental strategies, since efforts between authorities and polluters need to be harmonised (Skidmore, 2002).

In the last years, the main problems of implementing GIS are related to data availability and quality (Clarke *et al*., 2002). Lack of data and gaps in data coverage, accuracy and error propagation are listed as primary constraints for applying GIS. Also, data cost and data incompatibility are serious problems for researchers. Model applicability and development are referred to as the second major barrier for the efficient use of GIS in environmental applications (Bregt *et al*., 2002). In particular, scaling problems and validation of models appears to be troublesome for many researchers. Data problems could be overcome through better data infrastructure, standardization and accessibility. In modelling, there are no universal solutions. One could propose more focused models towards specific applications or, alternatively, the mass-use of GIS in environmental modelling. To overcome know-how problems and to get unique expertise, the best solution is inter-disciplinary work.

While GIS technology provides a plethora of techniques, and access to data is easing, our ability to define spatial patterns within given data is still largely determined by the relationship between the objects in the landscape, and the scales at which we observe them. There has been a considerable research effort in the field of setting monitoring programmes in the last two decades, enabled by the rapid development geographic information systems. Still, dependence upon the data model and processing, limits the comparability of the results of individual monitoring case studies and it represents a major drawback for the application of comprehensive environmental assessment.

Model development

The relationship between the real world and the data matrix, including spatial dependence, is influenced by the two phases of mapping from reality to any specific data matrix. Firstly, decisions taken on the choice of representation in terms of the representation of geographical space and the attributes to be included and how they are measured, and secondly, the accuracy of measurements given the chosen representation. The chosen representation constitutes the model of the real world that is employed. Any data matrix can be assessed in terms of the quality of the model; the first stage of assessment of a data matrix can be in terms of model quality (Haining, 2003).

Model quality refers to the quality of representation by which a complex reality is captured. It involves a finite collection of spatial objects, spatial relationships and variables. Assessment of model quality includes assessment of whether sufficient and appropriately measured data on the necessary spatial objects and variables are available and have been encoded according to the set of representation (Salge, 1995). The model quality corresponds to the following (Haining, 2003):

- Attribute representation
- Spatial representation
- Spatial resolution and aggregation

Decisions on how to represent attributes in geographical space are influenced by the specific application and the spatial scale of analysis. The choices could be made on the following grounds:

- Pragmatic approach: data are only available on a certain spatial framework; a common spatial framework for integrating different data sets is needed; this is the application chosen for this thesis
- Methodological approach: more powerful analytical methods available for some data representations than others
- Theoretical approach: 'right way' of capture the phenomena (Haining, 2003)

Data quality refers to the performance of the data set given the specification of the model. A far as the user is concerned both model and data quality affect a database's fitness for purpose, whereas model quality assessment is specific to the application; data quality assessment involves generic criteria (Guptill and Morrison, 1995).

In all cases, the sampling design is fundamental (Goodchild *et al.*, 1993). The way in which records are made in the field basically determines the potential use of data. Decisions on sampling strategy will strongly influence options concerning which analytical tools can be used. The selection of monitoring site, or area is the primary decision. In some cases the environmental problem determines the area, or the monitoring site is self-evident. More commonly, the study area is selected from some larger region. The selection can be done according to several ratios, by probability, by the researcher's intuition, for the representation of the larger region in some sense. Sampling could be stratified on the basis of certain assumptions. There are considerable variations in the size of the monitoring areas, the spatial and temporal resolutions, the number of different environments, the kind of raw data used.

An efficient allocation of resources requires a consistent concept of the handling and integration of data at defined working scales that are compatible with the operational levels of environmental planning and management. The basic presumption is that the broader the working scale the higher is the aggregation level of data that can be taken into account at the expense of thematic differentiation and geometric accuracy of the maps. On a fine scale at a local level, the need for the highest geometrical accuracy and sound differentiation of thematic map categories is evident in order to provide valuable environmental information and to comply with operational tasks. The density of the required information is high and mapping approaches have to include additional knowledge from other datasets. Since Estonian landscapes are very mosaic with high biodiversity, detailed information is required (Sepp, 1999). The geographical scale of analysis means that detailed estimates of spatial relationships such as distance or configuration are needed. In the given circumstances the choice may raise technical and conceptual difficulties.

Below I address a methodological framework for linking different modelling techniques and datasets at best efficiency by using GIS models in environmental monitoring.

Environmental monitoring

Environmental monitoring can be seen as a process by which we maintain an overview of the state of the environment (Wiersma and Wiersma, 2004). It provides essential data on how environmental 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.

In essence, there are an evolving set of basic principles for designing a monitoring programme. In theory, when developing an environmental 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 environmental indicators, times series). Only then should one carry out tests in pilot areas and later apply the methodology at a national level. In practice, every monitoring programme is unique, depending mostly on geographical coverage, landscape features, objectives of monitoring, available technology, and financial capacities. Often, programmes of landscape monitoring are policy driven (Groom and Reed, 2001). Monitoring programmes have to be feasible for the scale and for the scope of the survey in terms of objects and geographical coverage. Ideally, a monitoring framework should contain the following abilities (Clarke *et al*., 2002):

- The capacity to generate a representation of an environment at different scales
- Exhibit modelling and data processing capabilities
- Be object oriented and object specific
- Be mathematically feasible

- Be able to produce results that are spatially explicit and ecologically meaningful

Therefore, the monitoring programme should achieve information coverage in a cost-efficient and time-consistent practice. The resulting maps serve a wide range of applications in local planning, zoning, delineation of protection zones, design of management schemes and environmental impact assessment. For more strategic planning that goes beyond site-specific decision-making, a knowledge about trends in environmental conditions is an important source for authorities in order to elaborate guidelines for physical planning. The monitoring programmes should be more target- and problem-oriented, and the offered information and proposed measures more spatially focussed and more closely related to the mutual diversity of landscapes. For example, landscape monitoring focuses on specific values, i.e. properties of the 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 must in turn be adaptable.

Underpinning this research is a belief that many of the concepts used in environmental management are relative, i.e. the characteristics or qualities exist over a spectrum, and that discrete, binary categories such as high/low quality, natural/unnatural, clustered/dispersed fail to capture the fuzzyness of boundaries in space and time. For this reason this research is often focused on the identification and mapping of indicators and surrogates, and relative features to characterize the monitoring network.

Another problem is the complexity of monitoring subjects. Environmental quality is the aggregate of a whole series of more specific qualities, including, for instance, air and water, soils, landscape, biodiversity, waste and climate. Some of these qualities may be managed at a relatively local scale (nature reserves), others are more regional (landscape management); some are regional to international (air and water quality) and some are global (e.g. climatic change). Among the topics discussed below, landscape is a very complex phenomenon. Therefore, landscape monitoring methodology tends to become complex, covering various subjects, from biodiversity and vegetation monitoring through to 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 (Bailey and Herzog, 2004).

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 features such as soil, habitat, and water are monitored through independent studies. 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; Hertzog *et al.*, 2001; Peterseil *et al.*, 2004), 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; Fjellstad *et al*., 2001; Groom and Reed, 2001; Bailey and Hertzog, 2004).

Estonian national environmental monitoring programme

Environmental protection is associated with mandatory monitoring, management, and impact assessment duties imposed by Estonian law. Aiming at the maintenance of the Estonian environment, landscapes and biodiversity, conservation must not only capture in-depth information on status, management and impacts of the sites but has to include information on ongoing pressures affecting landscapes.

Estonia has experienced rapid changes in economic structure. As discussed in **Paper I,** industrial contraction, as the primary factor, has meant that emissions have decreased dramatically between 1990 and 1999. However, some Estonian point sources like oil-shale-based power plants are still reported to be among the biggest point source polluters in Europe. It is reason why the oil shale region of Estonia has been chosen as a one of areas for detailed survey. From an environmental perspective, the important question becomes to decide how effective abatement policies are in achieving real environmental improvements. Spatial analysis provides a good way of assessing the effectiveness of policies. In contrast, there is no strong indication yet that a spatial approach can form the basis of abatement policy practices. This could be a consequence of problems in the practical implementability of spatial instruments rather than in a lack of understanding of spatial relationships.

Efforts have been made to structure and manage national environmental monitoring activities since the early 1990s. Since January 1994, a National Monitoring Programme has been implemented in Estonia under the supervision and co-ordination 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).

The goals of environmental monitoring stated in law are: monitoring and reporting on the present status of environmental pollution, characterisation of environmental changes, monitoring of trans-boundary distribution of pollutants. Within the framework of the coordination of environmental monitoring in Estonia in 1999–2004, the author was involved in the facilitation of developments having substantial research consequences:

- development of a comprehensive monitoring data analysis system, and
- implementation of an operative monitoring system for environmental indicators and their variations in time.

The Estonian national environmental monitoring set of 11 monitoring themes incorporates around 1,700 monitoring stations, and reports a total of 227 parameters (Table 1). The temporal frequency of monitoring varies from continuous on-line to five-yearly, as determined by the dynamics of the phenomenon of interest. Several NEMP projects are related to the European networks, datasets, and regional projects in the Baltic Sea Basin and are founded on an international framework of standards, methodology and reporting.

Table 1. Structure and content of the Estonian national environmental monitoring programme in 2004

SUBPROGRAMME	No. of stations	No. of variables	Temp. resolution
1. METEOROLOGICAL MONITORING	**96**		
1.1. Meteorological	59	30	On-line
1.2. Hydrometeorological	37	11	12 per year
2. AIR MONITORING	**28**		
2.2.–2.3. Air monitoring in Tallinn and in Ida-Virumaa	6	8	On-line
2.5.Complex monitoring of air quality at Tahkuse	1	10	12 per year
2.6. Assessment of heavy metal deposition (bioindicators)	30	8	Annually
2.7.–2.8. Precipitation monitoring	18	15	12 per year
2.10. Monitoring of transboundary air pollution	3	5	On-line
3. GROUND WATER MONITORING	**499**		
3.1. Ground water monitoring	409	23	4 per year
3.2. Pandivere region monitoring	36	7	4 per year
3.3. Põltsamaa-Adavere region monitoring (sensitive areas)	19	7	4 per year
3.4. Monitoring of ground water organic compounds in north-eastern Estonia	10	11	4 per year
3.5. Microelements in ground water	25	18	4 per year
4. SURFACE WATER MONITORING	**175**		
4.1.–4.2. Hydrochemical and biological monitoring in Lake Peipsi	14	56	8 per year
4.3. Hydrochemical and hydrobiological monitoring in Lake Võrtsjärve	3	56	12 per year
4.4. Monitoring of small lakes	23	55	2 per year
4.5. Hydrochemical monitoring of rivers	62	21	12 per year
4.6. Biological monitoring of rivers	55	42	Every 5th year
4.7. Hydrochemical and hydrobiological monitoring in Narva	7	45	2 per year
4.8. Monitoring of dangerous substances	11	24	Every 5th year
4.9. Coastal processes in Peipsi and in Võrtsjärv	15	6	Every 5th year
5. MARINE MONITORING	**101**		
5.1. Monitoring of seawater eutrophication	45	24	8 per year
5.2. Monitoring of biota in the coastal sea	6	12	Annually
5.3. Monitoring of dangerous substances in the coastal sea	8	16	Annually
5.4. Coastal monitoring	42	9	Every 5th year

SUBPROGRAMME	No. of stations	No. of variables	Temp. resolution
6. BIODIVERSITY AND LANDSCAPES	**631**		
6.1. Coastal landscapes	26	5	Every 5th year
6.2. Remote Sensing of Landscapes	5	6	Every 5th year
6.3. Agricultural Landscapes	17	5	Annually
6.4.–6.15. Rare and Endangered Plant Communities (bogs, alvars, forests, meadows)	144	8	Every 5th year
6.16., 6.35–6.40. Birds	110	9	Every 5th year
6.24.–6.28. Rare Plant Species	304	8	Every 5th year
6.29.–6.30. Large Carnivores	8	4	Every 5th year
6.45. Soil biota	17	5	Annually
7. FOREST MONITORING	**116**		
7.1. Forest — I level	91	7	Annually
7.2. Forest — II level	7	24	Annually
7.3. Reforestation of mining areas	18	7	Annually
8. INTEGRATED MONITORING (Saarejärve, Vilsandi)	**2**	80	8 per year
9. RADIATION MONITORING	**26**	1	On-line
10. SEISMICAL MONITORING	**2**	10	On-line
11. SOIL POLLUTION MONITORING	**8**	24	Annually
IN TOTAL	**1684**		

As the case study of this dissertation, the Estonian national landscape monitoring programme concept introduced four monitoring sub-programmes: agricultural landscapes, coastal landscapes, protected and valuable landscapes, and land-cover by remote sensing (Table 1, 2 in **Paper III**). A draft concept of the Estonian landscape monitoring programme was presented to the Estonian Ministry of the Environment in 1995. To develop the Estonian monitoring programme, experiences from other countries were examined. “Landscape Monitoring and Assessment Research Plan” (O’Neill *at al*., 1994), “Countryside Survey 1990 series” (Barr *et al*., 1993; Bunce *et al*., 1993; Fuller and Brown, 1994; Howard *et al*., 1995) and LIM-Project in Sweden (Ihse and Blom, 1999) were assessed for the background, and aspects were incorporated into the Estonian plan. Since 1996, three programmes have been implemented (Sepp, 1999). 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.

For Estonia as new EU member country, the EU expansion required that Estonia meet the Copenhagen criteria, a set of standards designed to bring Estonian government operations and administrative systems into compliance with EU standards (SEC, 2002). A GIS approach was employed to assess the changes it had to make to meet the EU environmental requirements. To do so, the GIS based system was upgraded to assess monitoring policies and to select the most efficient and optimal way of meeting EU requirements. The GIS prototype includes data and modelling tools for the following purposes:

- to provide an overview of pollution sources, environmental systems, environmental quality conditions, as well as technical options for making improvements;
- to assess the changes to environmental quality that would result from implementing the various strategies, and plan the corresponding monitoring needs;
- to identify the low-cost strategies for meeting the legal requirements of the directives for environmental monitoring.

The prototype provided access to various databases and modelling tools through a GIS interface, which in turn offered a user-friendly method of specifying various scenarios. In order to present all networks of environmental monitoring within Estonia, the maps of monitoring sets for the entire country were generated directly from the prototype.

Critical issues in the field of environmental data management and presentation that remain to be tackled in the framework of this research are:

- to achieve data consistency avoiding overload and low quality data ('noise');
- to model controls and responses;
- to establish data filtering and query routines;
- to demonstrate evidence based impacts geographically.

2. DISSERTATION OUTLINE

Hypothesis and goal

This dissertation centres on the following hypothesis:

> Based on the inherent difficulties of dealing with conceptual incompatibilities in pattern analysis, environmental 'reality', and information demand, optimisation of environmental monitoring is needed. This optimisation compromises spatial representation of monitoring methodologies in the context of survey objectives, spatial and temporal scope, and availability of modelling techniques under chosen conditions. Presentation of an integrated approach for environmental information and mapping yields a spatial approach. To monitor larger coherent areas, a decision support tool is proposed that would enable efficient data management and design of monitoring networks.

Based on this hypothesis the main goal of the dissertation is threefold:

- ❑ To define a modelling strategy for designing and optimising monitoring networks/programmes related to the spatial impact of pollution.
- ❑ To provide a modelling methodology to assess the spatial importance of pollution sources, which addresses air pollution loads and compounding effects, and thereby to provide a means for assessing pollution policies and controlling instruments.
- ❑ To demonstrate modelling techniques for spatial structure and monitoring features in three case studies — modelling air pollution, landscape monitoring, and monitoring hazardous substances. These cases are applied within the Estonian national monitoring system.

The goal of the dissertation concerns the practical use and implementation of a proposed strategy to optimise the Estonian monitoring network.

Scope of environmental pollution

The spatial impacts of production on pollution output and load are complex. Pollution has compounding effects depending on the temporal and geographical patterns of the pollution load, on dispersion, and on deposition. The distribution of pollutants depends on the abundance and size of sources, on the spatial distribution of sources, and on the diffusion process (Heyes *et al.*, 1989). Their compounding effect embodies spatial and temporal dimensions. The spatial aspect is analysed in terms of scale, pattern (clusters or scatters), configuration (point, linear, areal) and human induced activities (number, type, magnitude). An additional complication concerns changes over time. Changes accumulate over time creating compounding effects. Implementing temporal GIS involves simple characteristics changing over time, as well as changing geographical entities (Longley *et al.*, 2005). If variations on a time axis can be measured, distinguishing artifacts and measurement errors and modelling 'true' change from normal geographical variability becomes an important issue.

The case study concerning modelling of air pollution includes dispersion modelling and a spatial module. The dispersion model used for this air pollution study in the north-eastern part of Estonia is based on the Gaussian plume diffusion model to estimate sulphur dioxide concentrations along an axis. A particular simplified solution of the full-scale diffusion equation is exploited. The trajectories of polluting compounds are calculated from the wind field at the level of the atmospheric boundary layer. Both long-term average and actual meteorological conditions are considered. A spatial model is employed in raster data structures. A sensitivity assessment of model parameters is undertaken to evaluate model structures and behaviour of parameters in different conditions. Finally, the utility of the model to assess environmental policies is examined.

Scope of environmental monitoring

The design of a monitoring programme is normally based on the methodology of a single study or theme, which sets research standards and represents geographical features of a certain natural phenomenon. The programme also needs to consider spatial and temporal variability. Comprehensive environmental analysis is impeded because monitoring sets usually do not overlap and in a relatively small sample the chosen set may be subjective due to practical limitations.

In practice, every monitoring programme is unique. Programmes of monitoring are often demand and policy driven (Groom and Reed, 2001). Monitoring programmes need to be feasible for the scale and scope of the survey in terms of objects and geographical coverage. The monitoring network needs to be optimised in both spatial and temporal scales, addressing:

- Appropriate data density and quality
- Efficient sampling strategies

A fundamental challenge when developing monitoring programmes is the selection of an approach for designing a set of monitoring areas. Theoretically, a random monitoring network is the optimal 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 Hertzog, 2004). However, depending on the selected monitoring variables, random monitoring often requires a vast number of monitoring sites and is thus quite expensive. Alternatively, a strategic approach using data collected for multiple purposes and an integrated and interdisciplinary approach are often chosen.

Scope of spatial modelling and spatial structure

Neighbourhood approach

In pattern analysis, spatial point patterns represent collections of entities whose geographical locations, rather than any quantitative or qualitative attribute, are of primary interest. A familiar example is a map of all trees in a forest stand, wherein the data consists of a list of trees referenced by their geographical location. The goal is to determine whether the points are more or less clustered than expected by chance (Greig-Smith, 1983; Dale, 1999). Clustering of monitoring networks needs to be referenced to the impact of pollution and to monitored phenomena.

One needs to get an overview of the environmental state of a location chosen at random or in the neighbourhood of some other location. The usual expectation is that values at adjacent locations tend to be similar. The importance of the neighbourhood relationships between the monitoring stations lies in the option to interpolate indicators based on the data of the nearest stations and in searching for a gradient or transfer function for the area. On the other hand, discrete phenomena, for example soil type or land cover, need categorical analysis. The density of sets could be weighed according to the distance zones and neighbourhood. For example, indicators computed at the landscape level yield relatively general information averaged over the entire landscape unit under investigation (Lauch, 2002). Integration of different monitoring environments and methodologies at different scales and sampling frequencies could create knowledge surplus and a substantial increase of efficiency in information management, avoiding overload and large quantities of low quality data. The importance of the neighbourhood relationships between the monitoring stations, the purpose of which is to achieve national coverage, lies in the following:

- The option to interpolate indicators based on the data of the nearest stations
- The distance dependence of impact factors
- Searching for a gradient / transfer function of the study area and identifying the spatial variability.

Objectives

A number of specific objectives have been established for this thesis:

1. To develop a modelling algorithm to simulate spatial effects in complex situations, to investigate sensitivity of outputs to variation in controlling parameters of the model and to examine the spatial importance of effects, and to assess management strategies for pollution control. A case study focuses on air pollution and its dramatic changes in the north-eastern part of Estonia in the 1990s (**Paper I**).
2. To describe the point patterns of Estonian monitoring networks, assess their neighbourhood, configuration, arrangement, and representation on selected criteria, to analyse the overlapping and coherence of the theme networks, and to allocate monitoring networks efficiently (**Paper II**).
3. To characterise landscape monitoring networks according to selection of monitoring concepts and methodologies. To assess monitoring networks, in particular their neighbourhood features, for integrated landscape analysis. To explore the option of data acquisition for multiple purposes from universally designed monitoring programmes, stated as a broader question in environmental management, environmental data acquisition and data mining, to support management schemes (**Paper III**).
4. To establish a monitoring network of hazardous substances in Estonia and in coastal areas based on collecting and evaluating all available monitoring data on priority hazardous substances in Estonia and in coastal areas. The objective of the case study is to give a comprehensive overview of the distribution of priority hazardous substances, persistent organic pollutants and heavy metals in Estonian waterbodies and in the coastal Baltic Sea (**Paper IV)**. A national monitoring network is proposed related to the distribution pattern of substances. The case study focuses on toxic priority substances listed in the Stockholm Convention and in UNEP Trans-boundary Air Pollution Convention protocols of persistent organic pollutants and heavy metals. In addition, the patterns of polychlorinated biphenyls (PCB) in grey seals (*Halichoerus grypus*) in the Baltic Sea are examined to establish a monitoring programme to consider trans-boundary impacts for the grey seal population in the Baltic. Distribution patterns of PCBs in the Baltic are compared with north-eastern and Eastern England and the St. Lawrence Estuary (Canada) (**Paper V**).

3. METHODS FOR OPTIMISATION OF MONITORING NETWORK

This part draws together the relevant methodologies that are applied in this research. The methodology of this dissertation comprises five parts, which are built upon a continuous principle: pollution modelling mapping, neighbourhood approach, methodology for integrated landscape monitoring and for monitoring of hazardous substances. First, at the beginning, there is a need to develop a methodology for modelling pollution (**Paper I**). The second, presented equally in all quoted papers, incorporates all elements and the nature of the mapping concept, and how this has been addressed. The third approach combines research methodology of a geographical fixed point related to its surrounding area, neighbourhood analysis (**Paper II**). Because of the limited application of neighbourhood approach in monitoring, this will draw on research that has illustrated some of the difficulties, challenges, and potential solutions of doing so. The fourth part outlines how the research builds on neighbourhood analysis to bring together mapping techniques and spatial statistics in a real attempt to map the essential core of landscape monitoring (**Paper III**). The outline models are presented below (Fig. 2). The outline is composed of 7 modelling steps from hypothesis and problem statement to the integrated analysis, spatial queries and solution proposal. The fifth elaborates the methodological approach for surveying the distribution of hazardous substances and for setting up a monitoring network for them (**Paper IV, V**).

The set of five methodologies allows us to focus on spatial extents of environmental monitoring to address focal aspects of the thesis on the explicit treatment of spatial heterogeneity and to encompass an essential part of the developments in current geographical information systems. Technological advances, including widespread availability of spatial data, the rapid rise in computational power and software to manipulate these data, contribute to the methodology for this research.

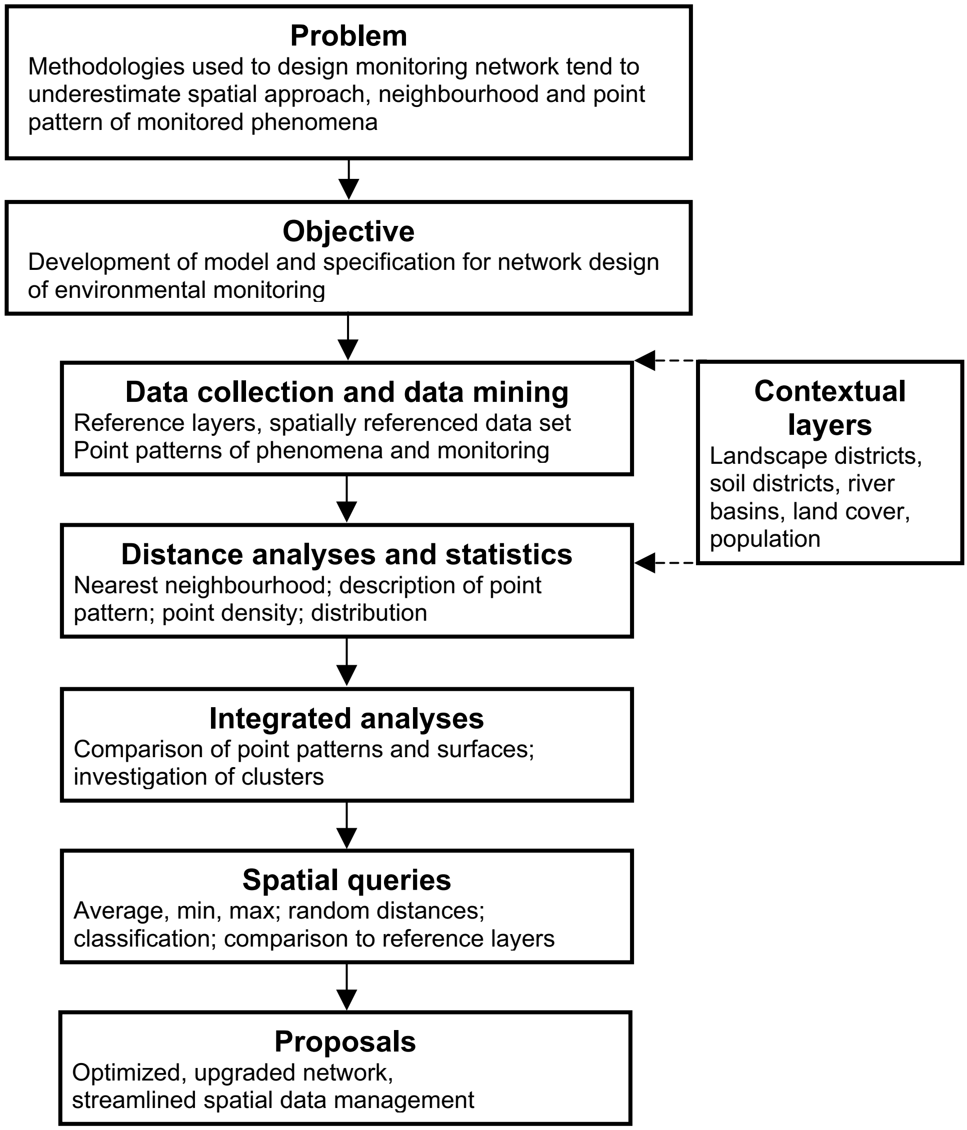

Figure 2. Outline of model

Pollution modelling

Compounding effect

The first part of the thesis, **Paper I**, focuses on air pollution, in particular modelling multiple impacts of various sources, examining their effect in different spatial and temporal resolutions.

Air pollution has compounding effect depending on the geographical and temporal patterns of load, on dispersion, and on deposition. The distribution of air pollutants depends on the number of sources of different size, on the spatial distribution of sources and on the diffusion process (Heyes *et al*., 1989). The spatial aspect is analysed in terms of scale, of pattern (clusters or scatters), of configuration (point, linear, areal) and of human induced activities (number, type, magnitude). Changes accumulate over time creating a compounding effect. There is an incremental effect of changes, called by Odum (1971) the 'tyranny of small decisions'. Theoretically we can distinguish two approaches of compounded effects. The first is just simple summing up of impacts,

$$\sum_{1}^{n} X = X_1 + X_2 + ... + X_n \qquad (1)$$

Another approach expresses compounding processes with probability functions. This simulates the complex behaviour of objects, or processes. In practice, the compounding effect of air pollution loads is the cumulative effect of a number of sources, which can be considered as a set of cases. In the modelling case in **Paper I**, 'a lot of polluters of the same pollutant' examines several polluters and one pollutant — SO_2. The geographical analysis is based on spatial dominance of a limited set of sources at different locations in the area.

Dispersion modelling

The model used in **Paper I** is based on the Gaussian plume diffusion model, estimating sulphur dioxide concentrations along an axis. The analytical expression represents a simplified formula of the full-scale diffusion equation. The trajectories of polluting compounds are calculated from the wind field at the level of the atmospheric boundary layer. Both long-term average and actual meteorological conditions are applied. Wind conditions are established using wind rose data. According to meteorological statistics the exponent of the wind speed profile in stability category D is 0.2. The model assumes that meteorological conditions remain constant during travel time, which is adapted for extended periods of release on an annual basis. Vertical profiles of wind and specific wind trajectories are excluded. If emission rate is large compared with the horizontal diffusion parameters, crosswind variations and Eddy diffusivity may be ignored (Clark, 1979). Sensitivity assessment of model parameters confirmed the reasonableness of these assumptions about model structure.

These assumptions provide the basis for a long-term, source-oriented model. Dispersion in neutral atmospheric stability conditions and the growth of a plume are described by dispersion parameters determined by wind speed and stack height in the following equation:

$$C_{ij}\ (r,z) = \frac{2Q}{r\, \alpha\, u_{sj}\, A_j}\ , \qquad (2)$$

where C_{ij} (r,z) is the concentration in a sector, i, for particular meteorological conditions, j, (g m^{-3}); Q is the emission rate (g s^{-1}); r is the distance from the source (m); α is the angular width of a sector; u_{sj} is the wind speed at the effective heights of source (m s^{-1}) and A_j is the boundary layer depth (m). The concentrations are valid for greater distances where the vertical dispersion coefficient is less than the boundary layer depth A_j. Methodological details about dispersion calculations are provided in Clarke (1979).

The results of dispersion modelling are imported to the spatial modelling module, assuming that pollution load in neighbouring cells is a function of the location of the cell in relation to pollution effects of surrounding sources. The final result of concentrations of pollutants is achieved by integrating, through simple addition, the results of the diffusion model across the entire study area according to source. The modelling exercise is carried out with Idrisi software. The OVERLAY module sums up sources in various atmospheric conditions. In time series analysis, two techniques, pairwise and multiple, are used to compare quantitative data images of air pollution. The outcome is the trends in change and the description of characteristic values and the abstraction of anomalies. The function of reclassification is used to divide the distribution into three classes, 2σ, greater than ± 2σ and values between these two, aiming to present 'true change'. The major database for empirical assessment is the national data on air pollution, supplemented by technological data of industrial units.

Sensitivity analysis

The value of the model is not to be judged solely in terms of predicted values. Equally important is the behaviour of the model output in relation to changes in values of the parameters and of the input data, i.e. its sensitivity (Suutari *et al.*, 2001). A small change in some parameters at one site may have a great effect on pollution fields, whereas the pollution may be relatively insensitive to changes in other parameters. As examined in **Paper I**, in each sensitivity test, one parameter is altered. The combined effect of changing two or more parameters at a time is also examined.

The analysis is concerned with a change in emission rate and in wind conditions. An analysis of the compounding effect on a regional scale disregards the influence of local circumstances and the sophisticated dispersion rules used in some models. Listed below are the input parameters altered for each test:

- wind direction — change of occurrence by 1% in 45° sector;
- wind speed — change of wind speed by 1%;
- emission rate — change of emission rate by 1%.

The initial part of the modelling strategy leads to a new set of research methods of spatial distribution, distinct from the principles above, which govern environmental pollution and dynamics of pollution load.

Thematic mapping

GIS has always provided for modelling a direct link to visualization tools and the lowest level of sophistication; GIS may be used just for mapping. Environmental maps, extensions in geographical representation, lead to improved planning and decision making, providing visual aids to researchers or managers (Goodchild *et al.*, 1993). Representation techniques could support the exploration of data sets and could draw out specific information for synthesis and dissemination. The GIS prototype model developed for Estonian environmental monitoring integrates eleven topics presented at national, district and local scales. The standardized approach for three basic map templates is applied through point, line and polygon data models. The selection of mapping tools, and upscaling and visualization techniques is given in the section below.

Thematic maps are produced as a co-result of modelling and end-user output. Rodriguez-Bachiller and Glasson (2004) set a number of criteria that thematic maps need to meet, including:

- Policy relevance in addressing the environmental issues and problems
- Analytical soundness — being based on sound science and mapping techniques
- Measurability in terms of data availability and cost effectiveness of data collection and analysis
- Interpretation — that the maps should communicate essential information in a way that is clear and easy to understand

Based on monitoring data, thematic maps are compiled using various modelling and mapping techniques (Slocum *et al.*, 2005). The following spatial operations are applied in the research:

I General operations of spatial databases:

- Storage of large amounts of spatially referenced information concerning an area in a relational database that is easy to update and use
- Rapid and easy display of visually appealing maps of such information, be it in its original form or after the application of database queries or map transformations

II Analysis in two dimensions:

- Map overlay, superimposing maps to produce composite maps, the most frequent case
- Producing partial maps containing only the features from another map that satisfy certain criteria
- Combining several differently weighted maps into more sophisticated composite maps, using map algebra
- Calculating size (area, length) of the individual features of a map
- Calculating descriptive statistics for all the features of a map, frequency distribution, averages, maxima, minima
- Carrying out multivariate analysis, such as regression, of different values of different attributes of a map
- Using minimum distances to identify features on one map nearest to particular features on another map

III Analysis in three dimensions:

- Using 3D model for density analysis of monitoring sets in regular grids

Three basic templates of map layout have been developed for mapping monitoring data: neutral base layers, depicting coastline, roads and rivers; insistent base layers, consisting of intensified borders; thematic layers, consisting of outstanding mapping features and symbols, most often for water environment or nature reserves. The presentation model of environmental maps consists of map layers in the following descending order: thematic layers; coastline; Tallinn (the capital of Estonia); relevant towns in counties; borders — county borders, state border; street layers — highways, state roads; hydrographical network; collar layer for borders.

Types of maps are specified for the cartographic presentation as follows: sign maps, proportional point and line symbol maps, isoline maps, delineation maps, area category, area distribution, point diagram maps and choropleth maps. Also, combinations of listed map types are applied in the set of environmental maps of Estonia. The cartographic modelling deploys the functionalities of Idrisi, MapInfo Professional® and Vertical Mapper. Featured examples of various environmental maps are presented in sections below and in quoted papers.

Three mapping scales were implemented as follows: national, district and local. Most commonly, monitoring activities involve the following three levels: monitoring at the site level, at a sample size of 1x1 km, focusing for example on the monitoring of habitat quality with reference to habitat profiles; then monitoring at the ecosystem level, focusing on functional and structural aspects of ecosystem dynamics in the larger context of adjacent environmental structure and processes; and monitoring at the landscape level, with special emphasis on wide-area land use aspects or environmental impacts. The most powerful layers, or indicative variables are identified and pre-selected by spatial variability. Developing indicator maps as end-user maps can help inform the policy making process of environmental processes that are important considerations for policy (MacFarlane *et al*., 2004). The potential application of the map as a tool could be for campaigning, for promotion of a local image, as a map on the wall for public presentation, map series for planning and for an environmental assessment application (Roose, 2002b; Roose and Saluveer, 2002; Roose, 2004b).

Although different techniques are available for data management and data mining, all thematic data is classified as point, line and polygon data models. Depending on spatial and thematic consequences some alterations are made by interpolation and aggregation. Then, rendering methods determine the appearance of objects on the map that emphasise certain aspects or objects in a model (Clarke *et al*., 2002). For point data at ordinal or higher measurement scale, quantitative point symbolization is applied. Point data is symbolized with the ordering visual variables: size, value (colour) and chroma (colour). Size, defined as the primary visual variable, is applied with range-graded symbolisation method, which generalizes data for better visual discrimination. To divide attribute values into classes, natural breaks classification, also known as Jenk's method, is often applied (Mapinfo, 2003). Alternatively, custom classification method is used if the setting for quality classes of featured variables is available. The value/brightness and chroma/hue of colours is applied to emphasize the graduation of data, considering that the choice of colours to be used in a visualisation affects the scientific utility of the visualisation and has a large psychological impact on the audience (Clarke *et al*., 2002). Colouring also allows discrimination of small changes in data values. For a temporal point data clustered column chart type is employed. If absolutely necessary, attributes are stretched to attain better visual distinction between different instants. Interpolation is applied for modelling continuous environmental surfaces. Interpolation aims to produce raster grids or isolines.

Contrast between base layers and thematic information is accentuated (Mapinfo, 2003). Therefore, the wide elements in insistent base layers are filled with less than 50% grey-scaling. To achieve distinctness and readability, all thin linear elements like coastline, borders and rivers are filled using highly saturated black or cyan. To emphasize thematic layers and textual elements, mainly in arial, halo effects are applied. Fully filled and more compact symbols are preferred in this environmental mapping project.

Environmental maps exploiting monitoring data provide comprehensive reviews and synopses. The set of maps addresses the current methodological and technical problems of environmental mapping, helping standardisation and the development of spatial data infrastructure and improving mapping tools. Following the framework of environmental monitoring, mapping techniques available, and constraints of data availability, the need for the following set of indicators maps shown in Table 2 has been recognised.

Table 2. The set of thematic maps

Subject area	Map set (no. of maps)	Map template	Techniques applied
Meteorology	8	Neutral	Sign map, isoline map, delineation map
Air pollution	20	Neutral	Sign map, shaded isoline, proportional point symbol
Groundwater quality	6	Water	Sign map, localised diagram, proportional point symbol, area distribution
Quality of surface water	29	Water	Sign map, proportional point symbol, line map
Quality of coastal sea	8	Water	Sign map, point diagram
Landscapes, soils	6	Nature	Sign map, area distribution, area category, choropleth map
Biodiversity	9	Nature	Sign map, proportional point symbol, area distribution, area category, choropleth map
Forest	3	Nature	Sign map, proportional point symbol, area category
Radioactivity	2	Neutral, admin.	Sign map, point diagram, choropleth map
Seismic	2	Neutral, admin.	Sign map, dot density

Neighbourhood statistics

Statistical techniques can be used to explore interactions between events at different length scales and to explore interactions between different types of events in the same area (O'Sullivan and Unwin, 2003). The methodology provides a range of statistical analyses for spatial point data, linking external models, as well applying internal functionality of the GIS packages. These include simple nearest-neighbour-derived tests and more sophisticated second-order statistics such as Ripley's *K*-function and the neighbourhood density function.

The spatial statistics in the dissertation are subdivided into four categories:

- Spatial distribution — statistics for describing the spatial distribution of incidents, such as the mean centre, centre of minimum distance.
- Distance analysis — statistics for describing properties of distances between incidents including nearest neighbour analysis, linear nearest neighbour analysis, and Ripley's K statistic.
- 'Hot spot' analysis — routines for conducting 'hot spot' analysis — hierarchical nearest neighbour clustering, K-means clustering.
- Interpolation — a single-variable kernel density estimation routine for producing a surface or contour estimate of the density of monitoring stations.

The primary task for environmental analysis is to define a statistically significant sampling size. The sampling designs and analysis techniques discussed below allow us to estimate the status of the environment in the given spatial extent and the magnitude of change. The establishment of sample-based monitoring programmes, collecting environmental field data integrated with thematic mapping projects, gives reason to expect the availability of a wide range of compatible, spatially referenced datasets in the future.

An approach using density field combines research into a geographical fixed point and its surrounding area. Object- and field-based models can efficiently coexist (Cova and Goodchild, 2002). The methodology, applied in **Paper II and III**, is based on the statistical description of point patterns and on neighbourhood analysis (Upton and Fingleton, 1985, 1989; Haining, 2003; O'Sullivan and Unwin, 2003). In this approach, the main features are distance parameters or distance statistics. The density of the monitoring network is weighted according to distance zones and neighbourhood, by assessing distances between points or the distance to geographical objects or factors. The point pattern of the monitoring network is analysed using Mapinfo Professional®, Vertical Mapper and CrimeStat (Levine, 2002). The thematic end-user maps are produced using Mapinfo Professional®. The basic data source is the national environmental monitoring programme. Other spatial databases such as the Estonian base map, Baltic Sea Region (BSR) database, environmental registry, CORINE land cover, population census, are involved in modelling.

Dispersion refers to the tendency for patches to be regularly distributed or clumped with respect to each other. Many dispersion indices have been developed for the assessment of spatial point patterns, some of which have been applied to categorical maps. One of the oldest and most proven 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; McGarigal and Marks, 1995). The distance to the nearest neighbour is calculated and averaged over all points.

$$d(NN)=\frac{\sum_{i=1}^{N}\frac{\mathrm{Min}\left(d_{ij}\right)}{N}}{\frac{1}{2}\cdot\sqrt{\frac{A}{N}}}, \tag{3}$$

where N is the number of points in the distribution, $\mathrm{Min}(d_{ij})$ is the distance between each point and its nearest neighbour and A is the area of the region. If the observed average distance is about 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 an indicator 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 monitoring sets of ground water and surface water, or landscape monitoring and soil monitoring.

The second function that is implemented for a model is based on Ripley's K-function. It is the 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, so 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 set.

Under unconstrained conditions, K is defined as:

$$K(d_s) = \frac{A}{N^2}\sum_i \sum_j I(d_{ij}), \tag{4}$$

$$d_s = \frac{R}{100}, \tag{5}$$

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:

$$L(d_s) = \sqrt{\frac{K(d_s)}{\pi}} - d_s \tag{6}$$

Spatial point data of the monitoring network for univariate analysis is stored in Excel as a list of *x*, *y* coordinates stored in two columns; a third column containing a numeric grouping code (e.g. specifying different strata, administrative areas, classes) may also be specified. Univariate data may be analysed as a single event set or a series of smaller event subsets based on the grouping information. Groups below a certain user-defined sample size threshold may be excluded from these analyses, as many of the tests used in this context are very sensitive to small sample sizes. For bivariate data two sets of *x*, *y* data are stored in four columns. Alternatively, a single *x*, *y* list (in two columns) may be stored alongside a third grouping column allowing specific groups to be selected for paired tests of association and the like (Perry, 2004).

The possibility of only being able to detect the pattern of impact arises if sampling effort is allocated in a highly asymmetrical manner. Allocating effort to estimate variability within sites at the expense of estimating among-site variability will greatly reduce the reliability of estimates of among-site variability (Cole *et al*., 2001). The greatest resistance to systematic sampling appears to derive from concerns about the independence of samples, and a lack of familiarity with its theoretical basis in statistical literature. In addition to the studies referenced in the Introduction, Cochran (1977) reviews systematic sampling, and Dunn and Harrison (1993) found that systematic sampling was highly efficient in a simulation study of land-use data, and that the post-hoc stratification of systematic samples still over-estimated the sampling error.

The spatial analysis of categorical data remains a critical issue, since it is little explored (Upton and Fingleton, 1989), though such information is often readily available, and may be cheaper to collect. One of the reasons for the lack of familiarity with the techniques may be the widespread availability of computer packages to carry out analyses of variance and neighbourhood statistics, but a lack of software for spatial autocorrelation (O'Sullivan and Unwin, 2003).

Methodological framework for landscape monitoring

The landscape mosaic model is chosen for this research. Landscapes are viewed as spatially complex, heterogeneous assemblages of patch types, which cannot simply be categorized into discrete elements such as patches, matrix, and corridors (Forman and Gordon, 1986; Turner *et al*., 2001). Rather, the landscape is viewed from the perspective of the organism or process of interest. The major advantage of the landscape mosaic model is its more realistic representation of how organisms perceive and interact with landscape patterns. The major disadvantage of the landscape mosaic model is that it requires a detailed understanding of how organisms interact with landscape pattern. The latter is out of the scope this dissertation.

In developing the landscape monitoring programme in Estonia, the landscape was defined as a regional unit, or geo-complex (Sepp, 1999). The key reference is landscape typology. Landscape unit provides a functional and methodological link between the bio-physical main structure and the socio-economic features of a landscape (OECD, 2001). Landscapes were considered as dynamic material systems formed by the interaction of substances and processes within the geo-sphere. Every landscape is inherently a geo-complex, in which a change in one component (plant cover or the water regime, etc.) affects the whole complex. If, in selecting study areas, a representative distribution according to the landscape regions is considered, interpretation of changes in landscape structures can be synthesized for a landscape district.

The importance of choosing the scales at which landscape indicators may be relevant as a tool for monitoring and management is the subject of academic debate within landscape ecology and monitoring. For international generalisation, the issue of regional differences within countries arises, especially with regard to the difficulty of developing national-scale average landscape indicators (Wascher, 2003; Peterseil, 2004). Most commonly, monitoring activities involve the following three levels:

- Monitoring at the site level (e.g. Natura 2000), at a sample size of 1x1 km and focusing on the monitoring of habitat quality with reference to 'habitat profiles' and on quality aspects of biodiversity related to human land use.
- Monitoring at the ecosystem level, focusing on functional and structural aspects of ecosystem dynamics in the larger context of adjacent environmental structure and process.
- Monitoring at the landscape level, with special emphasis on wide-area land use aspects such as agriculture, forestry, transport, and tourism, and the application of assessment for landscape change by sectors.

A fundamental question in developing a monitoring programme is the selection of an approach for designing a set of monitoring areas. A strategic approach using data collected for multiple purposes and stratified analysis is often chosen. National monitoring systems for landscapes and land use analysis are based on environmental strata.

Framework for monitoring hazardous substances and sampling

The third methodological approach aims to create a basis for designing a monitoring network for hazardous substances, in **Paper IV**. The Estonian case is compared primarily with the monitoring approach in Finland and Sweden (Ukonmaanaho *et al*., 1998; Heikkilä, 1999; Agrell *et al*., 2001; Voigt, 2004; HELCOM 2004b), however, comparisons with Latvian and Lithuanian cases are generally unfeasible (Klavins *et al*., 2000). It is essential for human health that all countries monitor potentially hazardous chemicals in food supplies. Many chemical contaminants are readily taken up by plankton, fish, birds, and mammals and become concentrated at the top of the food chain in marine mammals and fish (Wieder *et al.,* 1998; Roots and Zitko, 2004).

Designing the monitoring network of hazardous substances is incorporated in the drafting of water management plans. As a rule, planning is based on classification of water bodies, i.e. empirical data on their status. As the concentration of hazardous compounds in water bodies is usually below detection level, the focus of the monitoring is shifted to the biota and organisms.

The background of the monitoring programme of priority substances in the coastal sea, rivers and intakes is given in the publication Roose *et al.* (2003), including the list of sampled toxic substances. Four monitoring programmes of priority hazardous substances have been set up in Estonia:

1. Rivers: sampling fisheries, in particular,

2. Intakes: sampling sediments, as content in the water is extremely low,
3. Narva reservoir: sampling water,
4. Coastal sea: sampling biota.

The extensive data set comprises water quality data from the national water registry and environmental monitoring programme. Although data on organic pollution is often incomplete, with some years missing, it is sufficient for identifying spatial and temporal trends in rivers and in marine water composition, and the impacts on living organisms.

Chlororganic substances in fish and molluscs are sampled annually in three monitoring areas (Pärnu Bay, Tallinn and Kunda Bay). Consistent with recommendations of the Helsinki Commission (HELCOM, 2004b), the selected bioindicator is the female Baltic herring of two-three years of age. In the case of zoobenthos, only the content of metals is analyzed in *Macoma baltica* and *Saduria entomon*. Monitoring samples are collected once a year from three to five points in the southern part of the Gulf of Finland. In Estonia, heavy metal content in the ecosystem of the Baltic Sea has been surveyed since 1974 (Jankovski *et al.*, 1996), and comparable results originate from the second half of 1980s. In fish, the heavy metal content has been determined in their livers. Baltic herring have been caught in the autumn from the North-Eastern part of the Gulf of Riga as well as from Pärnu Bay and from the two areas (Tallinn and Kunda) in the Gulf of Finland.

Regarding the sampling in rivers, the concentrations of heavy metals are assessed in 15 Estonian rivers. The frequency of sampling of heavy metals is seasonal. Data-series for water quality began in 1992 and have continued up to today. In 1999–2001, the inventory reports of hazardous substances in intakes (sediments and water) were published separately for three Estonian counties: Lääne- and Ida-Virumaa (oil shale region), and Harjumaa (Tallinn and its surroundings), and jointly for all other Estonian counties.

Taking into consideration the results of inventories, a pilot programme for monitoring hazardous substances in intakes was launched in 2002. The design of the monitoring programme originates from the research methodologies and location specifics of a certain natural phenomenon of air and water environments, taking into account also the spatial variability of the phenomenon. The pilot sampling was split into three stages (Roose *et al.*, 2003). Sampling in the first pilot year focused on North-Eastern Estonia (Fig. 1B **in Paper IV**), in the second year, on the metropolitan area (Fig. 1A **in Paper IV**), and in the third year, on repeat tests and detailed surveys in undisclosed cases, areas and substances.

Analytical methods for hazardous substances

The Estonian Environmental Research Centre, where all the samples of the coastal sea and intakes programmes were analysed, is acknowledged by the German accreditation bureau *Deutshes Akkreditieruns-system Prüfwesen GmbH* (DAP) DAP–PL–3131.00 (2008–11–22). Description of sampling techniques as well as the analytical procedures for persistent organic pollutants (POP) can be found in Roots (2001). For quantitative determination of polychlorinated biphenyl (PCB) congeners, the internal standard IUPAC 189 was added. PCBs were analysed on a 90 m capillary column (DB-5) using gas-chromatography (Varian 3380) with an electron capture detector (ECD). PCB isomers with IUPAC numbers 28, 52, 101, 105, 118, 138, 153 and 180 were analysed. The detection limit for different PCBs was 1 $\mu g\ kg^{-1}$ fresh weight.

The analysis of heavy metals follows ISO 8288–1986 (E) (ISO, 1994). Analysis uses AAS VARIAN SpectrAA–250 Plus atom absorption spectrophotometer with graphite and flame furnaces. The analysis encompassed the evaluation of heavy metal levels in river and marine environments, in order to assess potential effects and to identify pollution sources. Detection limits are of great importance in the analysis of river water, as concentrations of heavy metals are very low. A higher detection limit may cause much higher load estimates than the actual load. Detection limits of two Estonian laboratories which are involved in sampling are given in table 1 in **Paper IV**. Long-term metal pollution in the Gulf of Finland has also been documented in bioaccumulation studies of the widespread bottom species, which exhibited elevated concentrations of Hg, Pb, Zn, and Cd (Ukonmaanaho *et al*., 1998; Sipiä *et al*., 2002). Having determined the amounts of heavy metals in rivers, in the liquid phase, and in the sediment, one can calculate heavy metal mobility, bioavailability, and toxicity values (Leivuori, 1998; Wieder *et al*., 1998; Toro *et al*., 2001). The methodology of ecological risk assessment is not applied in the dissertation, as the scope is purely to develop environmental monitoring.

Related to case study of seals in **Paper V**, the concentrations of the five 'common' PCBs in the seals blubber were scaled to a sum of 100, and the sum of their concentrations in mg per kg lipid weight was added as the sixth variable. The resulting data set (Table 1 in **Paper V**) was centred (mean of each variable = 0) and scaled (standard deviation of each variable =1). The set was then examined by Principal Component Analysis (Zitko, 1994) by the programme Matlab 5.0 (The Math Works Inc., South Natick, MA 01760, USA).

4. RESULTS AND DISCUSSION

The results and discussion section is organised, structured and presented in the following way. The first main section concerns the compounding effect of air pollution that was modelled for the north-eastern part of Estonia. This is followed by the neighbourhood analysis of Estonian monitoring network to explore spatial relations between network and environmental strata. Then, an integrated approach to design landscape monitoring in Estonia is elaborated. Finally, the data on hazardous substances is critically assessed and monitoring of hazardous substances is proposed. A discussion of the findings and their implications for monitoring policy and planning and commentaries on how this methodology could be taken forward to a national scale, reflecting in detail on ideas around environmental monitoring, is given simultaneously.

Compounding effect of air pollution and spatial sensitivity to changes

Inventory of sulphur dioxide emissions

The study area of this case study covers the north-eastern part of Estonia, where the industrial sectors are mainly based on oil shale. From the set of 75 point sources, 9 large sources, which dominate air quality with 96% of the district's SO_2 emissions, are selected for modelling. In the temporal elements of the models, SO_2 emission data for 6 reference years during 1980s and 1990s (1980, 1983, 1990, 1993, 1996 and 1999) are included. An emphasis was laid on updated data coverage of the 1990s, on an annual basis, characterized by the introduction of new air pollution standards, by structural change and by technological shift in the Estonian economy (Liblik *et al*., 1997; Liblik *et al*., 1999; Kimmel *et al*., 2002). A map of the major sources of sulphur dioxide emissions in the Estonian Oil Shale Basin for 1983–1999 is shown in Figure 1 in **Paper I**.

As seen from Figure 1 in **Paper I**, SO_2 concentrations have decreased gradually since 1983. To keep track of historical trends, 1980 represents the highest level of combustion of oil shale in the power plants, i.e. historically the biggest quantum of emissions, up to 240 000 tons of SO_2 annually. All sources show a clear decline during 1983–1990 due to a slow decline in the demand for electricity. In the 1990s, the transition markets and decreased demand have been responsible for a substantial decrease in SO_2 emissions. Gross industrial emissions decreased by 24% from 1990 (150 000 tons) to 1993 (115 000 tons) without substantial environmental measures. Emissions of SO_2 continued to decrease in oil shale power plants by 14% in 1993–1996 and by 17% in 1996–1999, mainly due to the decline in the demand for energy. However, this is also a results of several technological measures that were adopted. Improvements in combustion technology enforced by EU requirements were introduced from 1997. From 1999 net electric power output, accordingly emissions, has steadily increased.

Predicted change

The average concentration fields of the modelled SO_2 are given for reference years in the form of isoline raster maps. 99 and 95 percentiles are used to identify changes and to separate them from the errors inherent in dispersion modelling. The thresholds are taken as 3 or 2 standard deviations from the mean. Beyond these limits true change was interpreted. Table 3 gives the threshold data, which shows significant change at the 95% levels.

Table 3. Thresholding and areal impact of significance change

Statistics	Max	Min	μ	STD	Lower limit		Kappa Index of Agreement
Period	μg m^{-3}				μg m^{-3}	km^2	
1983–1990	0.71	–17.35	–0.93	1.25	–2.18	705	0.239
1990–1993	3.53	–25.83	–1.57	2.77	–7.11	150	0.218
1993–1996	7.30	–15.17	0.21	2.55	–4.89	56	0.688
1996–1999	–0.11	–8.77	–0.87	0.93	–2.73	189	0.239

Max — maximum change in SO_2 concentration; Min — minimum change in SO_2 concentration; μ — mean change; STD — standard deviation.

Proportionally, the biggest change in SO_2 concentration occurred between 1990 and 1993 (–1.57 μg m^{-3}). In turn, the change was weakest between 1993 and 1996 (0.21 μg m^{-3}). Variability of change (STD) characterizes the period of the most dramatic changes from 1990 until 1996, respectively 2.77 and 2.55. In general, areas experienced a significant decline in SO_2 concentration in an area of up to 705 km^2 around Narva in 1983–1990, having less impact in 1990s (up to 189 km^2 1996–1999). The Kappa Index of Agreement performs as the measure of difference between images of SO_2 concentrations (Rosenfield and Fitzpatrick-Lins, 1986). The Kappa index is largest for 1993–1996 changes, 0.688, when the largest area is affected, indicating changing meteorological conditions. Changes in terms of spatial and population impact are summarized in Figure 2 in **Paper I**.

Over the district there is a clear decrease in SO_2 concentrations, on average between 5–35% in 1990–1999. Most areas saw a decline of 15 to 25%. But, the populated areas saw a bigger decrease in SO_2 concentrations, in the range 20 to 30%. In particular, a large reduction is predicted by the model in the western part of the district, and between the core area and the eastern part. Changes in

SO_2 concentrations over the region are also given annually. For instance, the map of relative changes in SO_2 concentrations from 1990 to 1991 is polarized (Fig. 3 in **Paper I**). The reason for changes is the decline in emissions, but it is complemented by changes in meteorological conditions. The 1990s are exceptional in terms of climate which means that the inter-annual meteorological variability has to be tracked.

Liblik and others (1997, 1998, 1999) developed methods of estimating air quality, emissions of pollutants into the atmosphere, and the dynamics of air pollution fields for Ida-Viru county from 1960 onwards, including a detailed survey in 1995–1998 according to industrial units. Similarly to above presented results, their model shows lower concentrations of pollutants in the atmosphere in 1993–1995 and a worsening in 1995–1997, based on annual average concentrations. This work contributes by mapping changes and dynamics of spatial impact of SO_2 pollution instead of mapping pollution zones in annual repetition series. A major problem is that the model is unable to predict realistic concentrations around sources, to be discussed below.

Model validation and sensitivity analysis

The results of modelling are tested using the national monitoring network data in period 1990–1999. This data is obtained from measurements of SO_2 concentrations at four sites twice per day. All four sites are situated in towns. The validation of results is also done by comparing predicted concentrations with monitored data for the period 1991–1997 with available on-line measurements. Overall the relation between model response and observations is consistent. The model overpredicts observed values for about 20% of the mean observed values. Large overestimation is confined to areas close to the large sources, so the diffusion around the stack should be considered as a modelling error. Tendency to overestimate SO_2 in a radius of up to 5 km around a source became apparent. Therefore, the highest concentrations in the close neighbourhood of sources are adjusted to the level at 3 km from the source. Medium-distance diffusion equations allow such simplification. Secondly, the concentrations are disturbed because of impact of urban factors. The model is more reliable if there are fewer disturbing factors. Far away from sources, in the countryside, in the plain landscape the distribution of concentrations of SO_2 is smoother. At longer distances from the source, errors of multiple effect of sources are less compounded. Agreement is better around smaller sources. In rural areas, far from large sources, the agreement between modelled and observed values is most accurate, which directs attention away from traffic pollution.

The sensitivity of results is summarized in Table 4. The test shows that the model is most sensitive to wind speed, having an inversely proportional relationship with it. The wind speed is the crucial parameter, which dominates the model output. On the other hand, the sensitivity to wind direction change is lowest, just

+0.08%. The exercise of sensitivity assessment of emission rates was based one single pollution source, whose impact is described by a particular spatial impact function. In receptors, all sources with their impact functions were added iteratively; the mean change in modelled values is 0.8%. However, the spatial impact of emission change of a single source is crucial in air pollution mapping and is discussed comprehensively in the next paragraph.

Table 4. SO_2 concentration sensitivity tests

Test / Parameter	Mean change %	Mean modelled (1)	1–2 / 2 %
BASIC RUN (2)	–	5.50	–
Wind speed	–1.00	4.95	–10.0
Emission rate	+0.09	5.45	+0.91
Wind direction	+0.08	5.46	+0.72

The spatial impact function

The indicators of the spatial impact function for 1990–1999 in absolute and relative terms and as a regional total are presented below in Table 5. The spatial function is assigned to the policy analysis, since surrogates describe the importance of pollution load changes.

Table 5. Indicators of change in SO_2 distribution

Index	A	B	C	D	E	F	G
Source	Emissions change 1990–99		Relative spatial impact		Gross spatial impact		Elasticity
	Ind %	Rel %	Ind	Rel %	Ind	Rel %	F/B
Baltic PP	–53	39.3	0.390	36.0	–20.67	42.0	1.07
Estonia PP	–40	33.3	0.350	32.3	–14.0	28.4	0.85
Viru Keemia	–17	1.2	0.085	7.8	–1.45	2.9	2.4
Kiviõli	–87	3.3	0.041	3.8	–3.57	7.2	2.18
Sillamae	–46	1.5	0.024	2.2	–1.10	2.2	1.47
Kunda Cem	–95	7.5	0.027	2.5	–2.57	5.2	0.69
Püssi PP	–71	7.7	0.014	1.3	–0.99	2.0	0.26

A — change of emissions 1990–99, % ; B — regional weight of emission decline, %; C — rate of spatial contribution of regional SO_2 concentration estimated by impact of changed emissions; D — weighted importance of rates of spatial contributions, %; E — actual contribution to the change in regional SO_2 concentrations 1990–99, A*C, %; F — weight of actual spatial contribution of pollution load, %; G — elasticity, ratio of spatial contribution and emission change; Abbr: Ind — individual value; rel — relative (in district); PP — power plant; Cem — cement plant.

It is concluded that, by all measures, the most influential sources are two large power plants. The gross impact is biggest for the Baltic Power Plant (20%), leaving the Estonia Power Plant second (14%). The relative spatial impact of other sources is 4 to 8 times lower. On average, smaller sources control only around 2–3% of district-wide changes in pollution load. The gross spatial impact of the Kiviõli plant is two times bigger than its emission change, a condition that is termed elastic. The core area illustrates positive compounding effect while eastern areas show negative compounding effect in contributions to pollution loads. Most sensitive areas are areas of marginal impact of different sources, such as peninsula areas between the core and eastern parts of the region. Consequently, because of compounding pollution effect there can be a rise or fall in pollution depending on the location. The sensitivity to emission changes is also higher in the peninsula shaped area between Kiviõli and Kohtla-Järve industrial areas.

Referring to table 5, the spatial assessment sets relevant policy priorities to strengthen spatial impact of controls. The modelled sources of this study are listed among industrial units of the Integrated Pollution Prevention and Control policy. Authorities are to specify the agglomeration in the north-eastern part of Estonia at which limit values need to be set. This modelling exercise assists in this task, defining spatial impact of sources and assessing significance of the likely impacts on the environment by comparison with relevant standards.

Neighbourhood analysis of Estonian monitoring network

Neighbourhood indices

Nine monitoring networks, meteorology, air, groundwater, surface water, landscapes, flora, fauna, forests, soils were assessed through nearest neighbourhood indices. The key features of statistical analysis are given in table 6 below. In general, pollution related sets tend to cluster around 'hot spots', with few reference areas represented (Fig. 4 in **Paper III**). 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. Unlike the network for groundwater monitoring (Fig. 3), the monitoring networks of plant species and fauna are clustered in protected areas, in nature reserves. Compared with other sets, the meteorological monitoring sets are the most dispersed, according to the Euclidean measures, even over-dispersed (the average distance to the nearest station is 33.3 km; random nearest neighbour distance is 25.8 km). Similarly, landscape monitoring shows higher dispersion (the average distance to the nearest station is 25.1 km). It could be stated that, with less monitoring sites, the monitoring network needs to be more dispersed for broader geographical representation.

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 x 16 km. Estonia has 91 monitoring stations with 2,136 observation trees (Fig. 4).

Criteria for network design originate either from pollution source, human impact, or the intention to cover the nature reserves. Landscapes and soils tend to represent as many typological districts as possible; the meteorological network is characterised by the over-dispersed distribution of stations to explore opposing marine-continental features. Consequently, regular, dispersed, aggregated and random patterns are observed in the Estonian monitoring set.

Table 6. Nearest neighbourhood statistics of monitoring network

Monitoring network	Stations: sample size	M: Mean Near-Neighbour Distance (km)	R: Mean Random Distance (km)	M/R: Nearest Neighbour Index	Distribution Pattern		Key factor for the network design
					Clustering	Randomi-sation	
Ground water	464	1.19	6.56	0.18	h. clustered	aggregated	pollution
Soils	51	7.83	15.31	0.51	clustered	random	coverage by regions
Plant communities	127	8.11	12.07	0.66	clustered	stratified	reserves, Natura 2000
Fauna	124	8.51	11.81	0.68	clustered	stratified	reserves, Natura 2000
Air	24	19.76	26.49	0.75	clustered	stratified	pollution
Rivers, lakes	82	13.36	14.37	0.99	neither	stratified	pollution
Forest	91	16.90	14.55	1.16	dispersed	regular	forested areas
Landscape	45	25.09	21.42	1.17	dispersed	stratified	coverage by landscape types
Meteorology	29	33.31	25.83	1.24	over-dispersed	stratified	marine

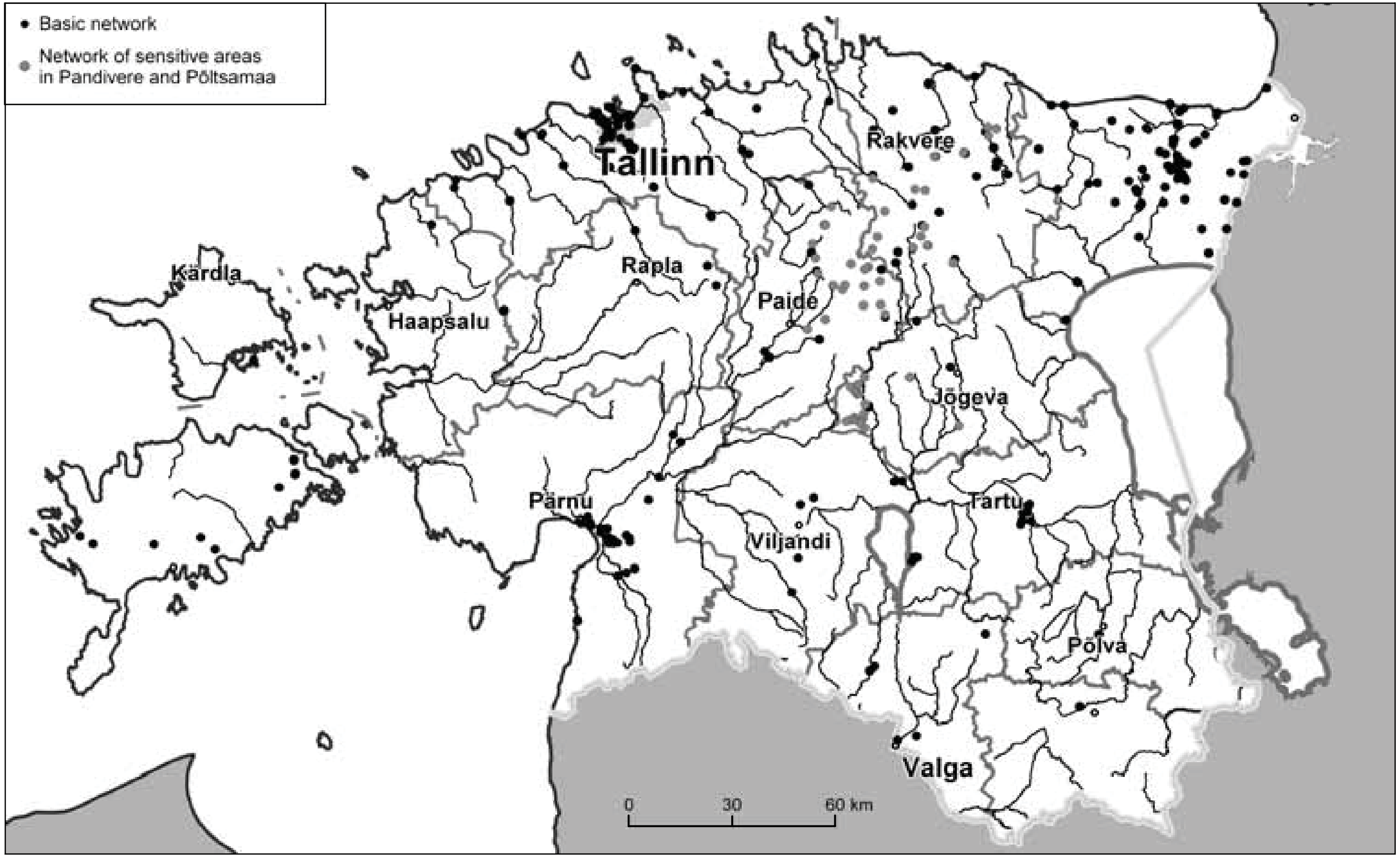

Figure 3. The network of groundwater monitoring and 'clusters'

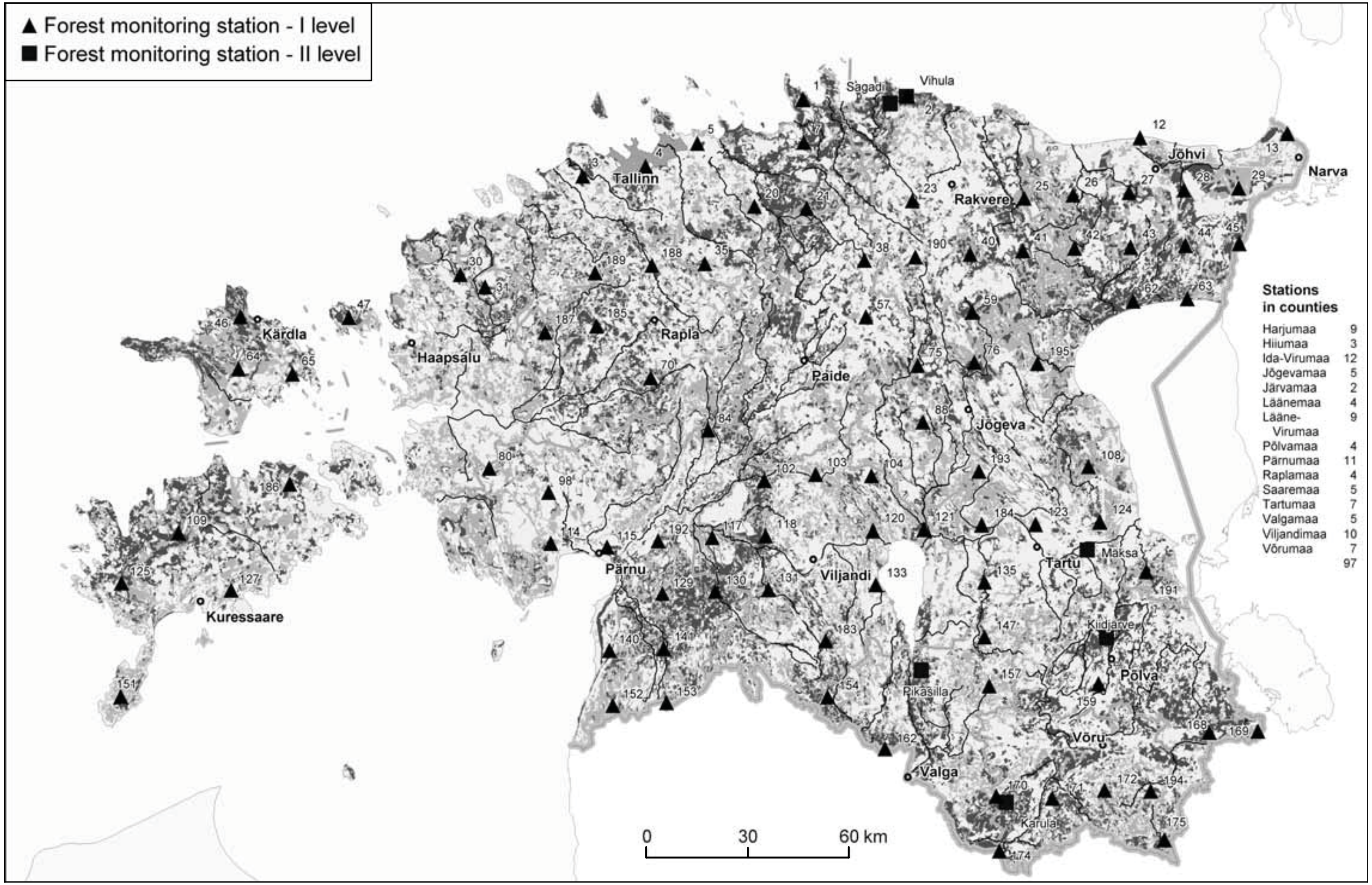

Figure 4. The network of forest monitoring, 16 x 16 km grid

Explaining the curve of k-order nearest neighbour indices (Fig. 4 in **Paper II**), 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, namely the Tallinn cluster and the cluster in north-eastern Estonia. Paired sites obviously describe the distribution pattern of landscape and inland water monitoring sets. In general, regarding all topical sets, after the fourth rank nearest neighbour, the differences between sets become less pronounced.

Ripley K-function describes the hierarchy of clustering (Fig. 5 in **Paper III**). Clustering is expressed clearly in the groundwater monitoring set, having a radius of 30 km for groundwater bodies (Fig. 3). Hierarchical clusters are clearly described in the plant species monitoring set, where the density of the point pattern increases at 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 the increase of clustering over long distances. According to Ripley's function, monitoring of fauna and forest sets are random and dispersed over longer distances.

The distribution of L between the sets and for various baseline characteristics such as distance from the sea, distance from a city has been compared in the analysis. Distance tests, shown in Fig. 5, indicate that the monitoring network for landscapes, plant communities and air pollution are marine. Regarding the distance from cities, treated as the distance indicator for human impact, wildlife-oriented monitoring sites are at a mean distance of over 15 km; air, inland water bodies, and meteorology networks are obviously more exposed to population.

The environmental monitoring network in Estonia has 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 total topic-based monitoring may be considered incidental, because the sets of the sub-programmes are independent of each other. On that assumption the total density map of monitoring network is derived by summing up 9 topical maps, the distance dispersion model at a 50 km search radius, shown in Figure 6 in **Paper III**. According to the model, stratified information on a multitude of themes is provided 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 unmonitored areas are delineated in northern Kõrvemaa, in Avinurme, and around Varbla. Consequently, all these areas belong to bigger landscape districts in sparsely populated forest and rural areas. As a density map has a linear dependence on the width of the search buffer, the methodology of the monitoring and the spatial function of the environmental phenomenon should be considered. Regarding the 50 km search radius, it was assumed that a transfer function is applicable for such a distance in many cases.

Also, 50 km could be taken as the average maximum distance between the nearest-neighbour monitoring stations.

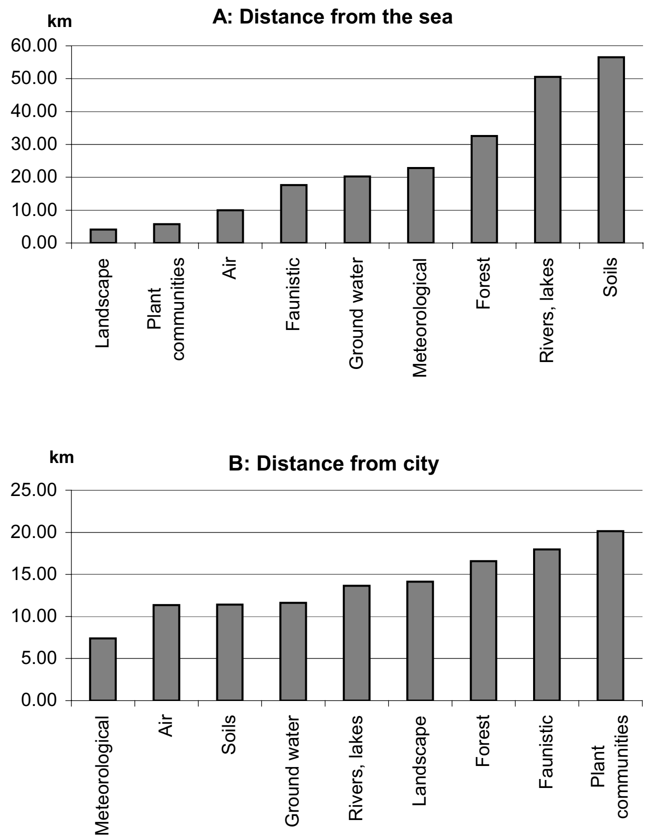

Figure 5. Distance indicators for the network of environmental monitoring in Estonia — A: Median distance from the sea, B: Median distance from a city

Designing a monitoring network to be spatially more feasible, related and coherent 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 place stations where there is no monitoring to follow the representation of phenomena. The histogram of the total density map of monitoring networks could reveal the patchiness of coverage. In a relative comparison, the environmental data coverage could be filtered by land cover or population density maps to highlight general discrepancies. The discussion how the design of a monitoring network follows neighbourhood statistics is given below via two case studies, a case study of landscape monitoring and surveys on hazardous substances.

Integrated approach to designing landscape monitoring in Estonia

In this section, the distribution of monitoring sets is assessed, aiming to set up criteria for the representation of landscapes. Monitoring sets are geographically analysed by land cover, by landscape districts and by soil units. Also the representation is assessed on regular sets like the Estonian square kilometres database and UTM grids. The examples presented below demonstrate that categorical analysis and mapping is a useful and informative technique, having potential value for decision-making and for integration of environmental stratas.

Classifying a large area into a number of regions that are relatively homogenous for some set of landscape attributes provides a useful framework for focussing attention, summarizing patterns, aggregating information, allocating resources and priorities in environmental management. These regions can be interpreted as vegetation types, land cover, landscape or environmental domains, depending on the attributes chosen, the map scale, and the objectives of the analysis. Environmental attributes can include climate, geology, landscapes, soils, vegetation, flora and fauna. The premise for such a synthesised approach is that physical environmental processes drive ecological processes, which in turn are responsible for observed landscape patterns and associated patterns of biodiversity.

In landscape monitoring in Estonia, a strategic approach in the selection of monitoring areas has been promoted. The main criteria, preconditions for the selection of stations and sites, are as follows:

- distribution according to Estonian landscape districts;
- distribution throughout the country;
- intensive and extensive areas as well as marginal areas of agriculture;
- availability of additional data;
- relation to other environmental monitoring sites, especially to biodiversity monitoring network.

Monitoring of agricultural landscapes is supported by datasets of environmental monitoring. The 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 in 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 strategic approach in the selection of monitoring areas based on landscape districts is cost-effective but has its own limitations in the interpretation of the results. It seems that the set of agricultural monitoring areas may not be sufficient for summarizing monitoring results per landscape district. Either we should increase the area of monitoring sites or increase their number. At the same time 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 (Roose *et al*., 2005). Additional 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.

Distribution of network for landscape monitoring

According to the first mapping of 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 (excl. 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, followed by 168 stations on land principally used for agriculture, and 129 stations in cultivated fields (Table 3 in **Paper III**). 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. For example, assessing the representative distribution of a network of plant communities, broad-leaved forest, coniferous forest and mixed forests are represented disproportionally well in relation to their areal coverage. On the other hand, modest exposition of the monitoring of plant communities in agricultural fields is justified by the intensity of land use, simply, there are no natural communities to monitor. In contrast, the purposeful location of monitoring sites is demonstrated by the intensive monitoring network in natural grasslands, beaches and dunes, mineral forest, swamp forest, marshes and raised bogs in order to track changes. The monitoring set of plant communities intensively covers alvars in coastal lowlands. Natural grasslands are proportionally over-represented due to the targeted monitoring of rare and endangered species. In the case of faunistic monitoring, the network is not widespread in wetlands, contrary to the assumption of extensive monitoring of wildlife in these areas. To explore the representation of land cover classes, deeper ecological knowledge and additional data for evaluation are needed for each spatial stratification.

The set of forest monitoring corresponds more or less to a proportional random selection throughout different forest types. 51 stations (59%) are located in coniferous forest, 27 stations (31%) in mixed forest, and 9 (10%) in broad-leaved forest. According to CORINE land cover (Meiner, 1999), coniferous forest covers 36% of forested areas, mixed forest, 35%, and broad-leaved forest, 16%, leaving 13% for transitional forests. As a result of comparison of representation by forest types, the focus in monitoring is given to coniferous forest, stressing the importance of this type in both forestry and landscape preservation. In forest management, coniferous forest is of primary commercial interest. Representation of monitoring stations in mixed forest is proportional to the coverage of this type of forest in general.

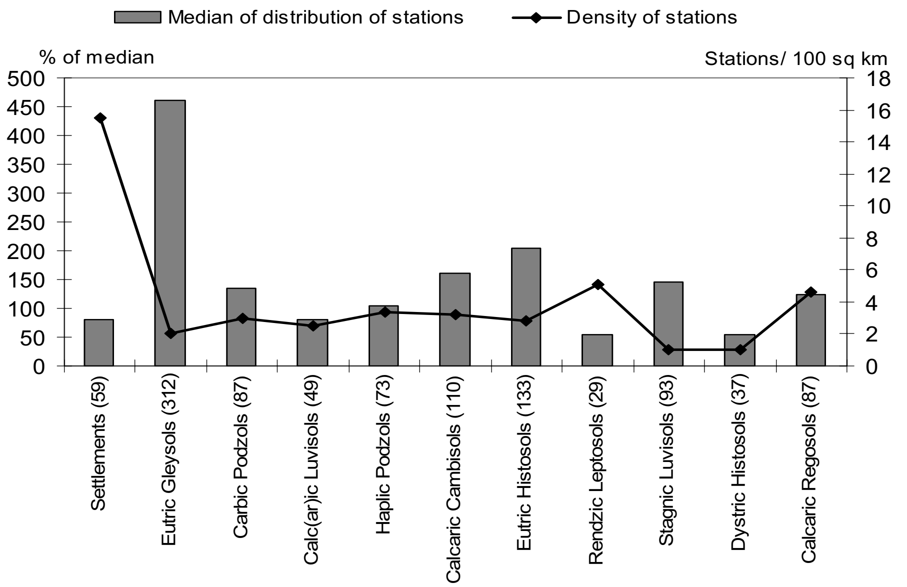

Figure 6. Distribution of monitoring stations and their representation according to soil units (no. of stations in brackets)

Examining distribution of monitoring stations by soil units (Reintam *et al.*, 2000), rendsic leptosols and calcaric regosols are over-represented (Fig. 6). On the other hand, stagnic luvisols and dystric histosols are under-represented. The distribution of stations by soil units is related to the distribution by landscape districts, since soils as a layer determine landscape classification. Large landscape districts are proportionally less represented in the overall monitoring set, and small districts such as the North-Estonian lowland, Karula upland, and Palumaa are more intensively surveyed (Fig. 2, 3 in **Paper III**). The existence of a dense network in Karula upland, means that there is tendency of lower representation in neighbouring districts, in Haanja, Otepää and Võru-Hargla. According to geographical distribution, lowlands are intensively covered by monitoring network.

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 districts having the same land cover or soil types enhances data mining techniques and best suits the available sampling set for the research objectives. Classification of data by geographical attributes improves the ability to exploit common object- and field-based analysis functions.

Designing the monitoring network

There are several principles of sampling and spatial representation in the monitoring programmes. In the case of endangered species of I and II category (for example, endangered vascular plant species, eagles), all habitats are present in the monitoring programme. Typological representation is chosen in the monitoring of the coastal landscapes. The large choice of criteria for the selection of monitoring sites affects the monitoring of agricultural landscapes. Comparative selection, reference areas versus impact sites, is the framework for setting monitoring stations for rivers and wildlife habitats. Also, as the statistical basis, the geographical normative 'density of network' could be applied to set the range of the sampling required by Eurowaternet in lakes and for groundwater.

In practice, it is not always easy to define the spatial representation and adjacencies of pre-established methods of monitoring and information needs (Pooler, 1992). Importance of neighbourhood consists of extensive data coverage, generation of data fields by interpolation and extrapolation, possibility to search the transfer factor for variables, exploration of variability and an understanding of the spatial structure of environmental features. The larger the standard deviation, the more extensive is the monitoring needed for that variable. Hence, it is obvious that the point pattern of monitoring differs from environmental features. Once one comes to some agreement about measures of scale and complexity, it will give the opportunity to move forward significantly in comparing data and models. The transfiguration of monitoring pattern can not exceed the range of natural variability of the monitored subject. Also, differences in the extensity of surveying play a role in the design and planning of monitoring networks.

Advanced modelling techniques allow us to map the diffusion of the impact of monitored variables over space. Therefore, it has been possible to produce continuous surface maps of relative monitoring data coverage, rather than zones of high/medium/low monitoring intensity (Fig. 6 in **Paper III**). The mapping of monitoring uses a single statistic plus a variation of distance variables from stations. Maps reveal areas where it is more likely to get monitoring information, but they do not produce sharp lines dividing monitored from non-monitored areas. Intensively monitored areas are those which have higher overlapping of environmental theme strata, hence with a higher cumulative value. Still, the discussion of threshold levels is generalized. The mapping exercises take varying conditions into account, notably distance to urban areas and to the sea and vegetation, and landscape features. However, there is limited consideration of interactions between strata and factors and how they may affect the total monitoring map, since, in general, the correlation between density layers is quite low. 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 locations of the monitoring stations, but also look at the monitoring task as a whole.

Maps are decision support tools, and final maps from a complex process communicate significant findings from the preceding research. Maps have a clear value as a tool, having the potential to be visually impressive, attractive and attention-grabbing and to communicate a great deal of information through a graphical medium. However, in the main, maps are the end product of a process. This research has aimed to develop a process for the assessment of monitoring networks and mapping, however, although the maps are in no way incidental to the research, any application of the mapping process must be careful since the map product could be erroneous or misleading. From a planning and management perspective, the disaggregation maps, a series of network-based classification maps, draw attention to which network is optimal, which could be filled in. Figures 3, 4, 7 in this chapter and Fig. 3, 6 in **Paper III** provide a set of examples of map applications of the overall model.

In the following attention is paid to the process of creating a grid onto which the monitoring data will be interpolated and extrapolated. In optimising the monitoring set, different location-based models have been applied. Uniform grids are the easiest and simplest type of grid. A regular grid as the basis for classification is chosen in the case of the Estonian square kilometres spatial database and for dispersion by UTM grid (Fig. 7). The representation in the Estonian square kilometres database considers random representation. The nationwide environmental topical layers have been compiled in the Estonian square kilometres database by Remm (2002). In this context, the national monitoring network embraces 944 km^2, which is just 2% of the Estonian territory. 666 grid squares, 70%, contain a single monitoring site, while 146 squares contain 2 stations, 86 squares contain 3 stations, and just 31 squares compound 5–9 stations. In fact, straightforward integrated multi-topical data coverage is available only in these grids.

Meso-scale assessment of network distribution is modelled three-dimensionally in UTM 10 km grid, which is the basis for several cross-European biodiversity studies. The 3D model shows that extensive monitoring grids are situated in the surroundings of Tallinn, Kohtla-Järve, Pärnu and in western Saaremaa (Fig. 7). The only overlapping with intensive monitoring areas occurs in western Saaremaa, in the Vilsandi nature reserve. The importance of other intensive monitoring areas in the Soomaa and Karula national parks, and in Saarejärve integrated monitoring site, is not expressed in the number of stations. The 3D model of the monitoring network for UTM 50 km grid is even more expressive; in addition to the above listed areas, grids in western Virumaa, Tartu and the south-eastern part of Estonia appear as accented monitoring areas.

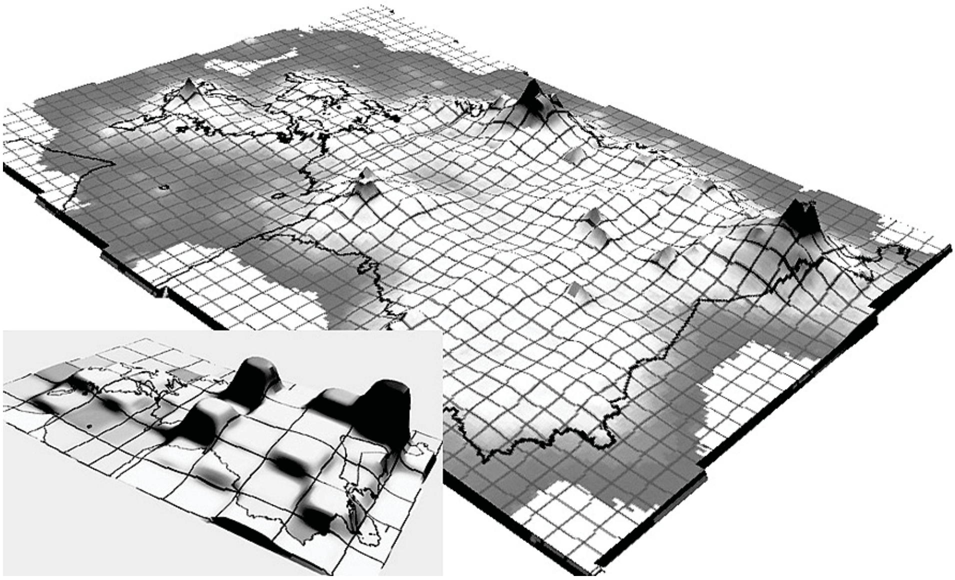

Figure 7. 3D model of monitoring stations in 10 km and 50 km UTM grid

Here, the discussion on methods and approaches continues. In recent years and definitely in the near future the monitoring network will be expanded due to advances in methodology and techniques (Folving, 2001; Bastian *et al.*, 2002; Haines-Young *et al.*, 2003; Groom, 2004). It is arguable whether the data of the national monitoring network is sufficient and cohesive enough for calculating any Estonian averages, although these are generally used as indicators, or indexes in European reports. Do we assume representation by the typological classes, or do we fill in the pattern with ‘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 multiple phenomena make 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.*, 1994; Dramstad *et al.*, 2001; Lausch and Hertzog, 2002; Opdam *et al.*, 2003). 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, in the case of landscape surveys (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 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 areas delimited according to ecological criteria (landscape districts, catchments, vegetation, 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 interactions with the existing administrative bodies. Whether they are related to eco-regions or administrative districts, data management needs to be harmonised.

Setting monitoring network of hazardous substances in Estonia based on their distribution

Although a lot of information is available, as stated in **Paper IV**, a comprehensive overview, for example, of priority hazardous substances is not available due to lack of data management, cross-national synthesis, and integrated framework projects in this field in Estonia. Reasons for the trends of last decades: in majority, declining concentrations of hazardous substances, are not fully understood. Also, data and discussion in the wider geographical context of the Baltic Sea Basin as a reference area is needed.

This part of the research, developed in **Paper IV and V**, aims to evaluate all available monitoring data on priority hazardous substances, assess their distribution in Estonia and in the coastal area, and accordingly to design a monitoring network for priority hazardous substances. The focus is given to toxic priority substances that are listed in the Stockholm Convention and in United Nations Environmental Programme (UNEP) Trans-boundary Air Pollution Convention protocols of persistent organic pollutants and heavy metals. The protocols require a ban of or the minimisation of these priority substances (UNEP 2003). Persistent organic pollutants (POPs) and heavy metals (HM) are a group of toxic and persistent chemicals whose effect on human health and on the environment include dermal toxicity, immunotoxicity, reproductive effects and teratogenicity, endocrine disrupting effects and carcinogenicity (UNEP 2003). The import of chlororganic pesticides to Estonia was prohibited by a government regulation from 1967. Estonia itself has not manufactured chlorine organic pesticides at all. Map of monitoring set of priority substances of the Estonian National Environmental Monitoring Programme (NEMP) is shown as Figure 1 in **paper IV**.

Persistent organic pollutants in the Estonian coastal waters and in rivers

The distribution of polychlorinated biphenyls (PCBs) in the surface sediment reported in HELCOM (2002) suggests that 'hot spots' have been identified in several locations, but not in the Estonian coastal sea. The highest level of PCB contamination was observed in the eastern Gotland Basin and in Lübeck Bay. Higher concentrations were also found near Stockholm, Viipuri and Klaipeda. PCBs had entered the environment in large quantities for more than 37 years and were bio-accumulating and depositing in sediments (Koppe and Keys, 2001). The role of long-range transport dominates in Estonia and its coastal sea, though the interaction of (airborne) POPs with surface media is not sufficiently understood (Scheringer *et al.,* 2004).

The concentrations of DDT and PCBs in the tissues of Baltic herring have decreased during the years 1995–1998, but there has been a certain rise during 1998–2001 (Fig. 2 in **Paper IV**). However, the reasons for the increase are unclear. It is possible that DDT has recently been used and discharged from Latvian or closely adjoining territories (Olsson *et al*., 1999). In the area of the Baltic Sea during the period 1994–1998, the highest DDT and PCB concentrations in herring muscle tissue were found near the German coast. The lowest PCB concentrations were found along the Estonian coast, but also in the northern Bothnian Bay and in the Kattegat (Olsson *et al*., 2002).

European Union (EU) Council Regulation 2375/2001 put the threshold limit value of PCDD/Fs in fish at 4 pgTEQ g^{-1} wet weight. The Baltic Sea fish have been separately highlighted because, in terms of PCDD/Fs content, they may presumably exceed the threshold, in particular for older Baltic Sea herring. The comparison of dioxin concentration in muscle tissue in Estonia and in Finland shows statistically reliable correlation between concentration of dioxins and age of fish ($r > 0.8$) (Fig. 3 in **Paper IV**). Concentrations of HCH-isomers (lindane) in water and biota have decreased considerably since the early 1980s. Concentrations of dioxin and PCBs in marine ecosystems declined in the 1980s, but this decrease levelled off in the 1990s. Dioxin levels in fish still exceed the new EU food safety limits in some areas, particularly further north in the Baltic Sea. Concentration levels of POPs are still so high that they have potential biological effects, at least in the Kattegat, the Belt Sea, and the Sound. The conditions differ substantially between the Baltic Proper and the Estonian coastal sea. The dioxin congener profiles in the Estonian coastal sea from herring in the western Gulf of Finland are similar to those from the central Baltic; those from the middle of the Gulf of Finland are similar to those from the Gulf of Riga (Roots and Zitko, 2004). Of the twelve Baltic herring samples taken from Estonian coastal waters and the Central Baltic, the dioxin content of only one of them (a fish older than 6 years and more than 17 cm in length from the Central Baltic) was above the internationally permitted threshold. For other endocrine disrupting substances and new contaminants like flame retardants, a full assessment of their levels or effects is not possible due to the lack of monitoring data.

Using the total water discharge into the Baltic Sea via rivers (475 km^3 y^{-1}) and using the median concentrations of 0.7 ng l^{-1} PCBs, 0.06 ng l^{-1} of DDTs and 0.1 ng l^{-1} of HCHs, river transport results in an annual quantity of 332 kg of PCBs, 2.8 kg of DDTs, and 47.5 kg of HCHs to the Baltic Sea. At the beginning of the nineties, the rivers and the atmosphere contributed about equally to the PCB load in the Baltic Sea, while for pesticides, atmospheric deposition was about 5–7 times more important (Fig. 6 in **Paper IV**) (Agrell *et al.,* 2001).

The survey in the target areas near Tallinn and in the oil-shale region of Estonia indicated (Table 3 and 4 in **Paper IV**) that the concentrations of all tested pesticides in sewage water was below target levels. Also, the concentration of aldine, dieldrine, endrine, DDT, hexaclorocyclohexane and hexachlorobenzene in the sediment samples was below the permitted target levels. The concentration of POPs is below detection level. The main reason for this is the dramatic decrease in the use of agricultural chemicals in Estonia in the 1990s.

Distribution of polychlorinated biphenyls in grey seals

Seals were chosen as the subject of a complementary case study because they live at the top of the food chain of the marine ecosystem, and accumulate many highly toxic compounds. Grey seals are mostly concentrated in the West-Estonian Archipelago Biosphere Reserve (Fig. 2 in **Paper V**). The reserve is one of the best breeding areas for seals in the Baltic Sea.

The concentrations of toxicants in the seals of the Baltic Sea have been surveyed earlier (Blomkvist *et al*., 1992; Haraguchi *et al*., 1992; Roots, 1996, 1999), but the results are not compared with the data and results from other regions. In fact, the feeding habits of the ringed seal include ingestion of major quantities of benthic crustaceans that might cause the observed differences in retained PCBs, whereas grey seal feed mainly on fish (Söderberg, 1975).

The profile (percent in mixture) of polychlorinated biphenyls (PCB) 101, 118, 138, 153, and 180, and the sum of their concentrations in mg kg^{-1} lipid in grey seals (*Halichoerus grypus*) from the Baltic, North-East and Eastern England, and the St. Lawrence Estuary in Canada, were examined by Principal Component Analysis (PCA). The first three principal components accounted for 86% of the original variance. As can be seen from Figs. 3–6 in **Paper V**, there are no 'clusters' in the data. However, the data contain several 'outliers', such as the samples 8, 10, 21, 25, 26, and 30 (Table 1 in **Paper V**). The reasons for the 'outlier' status can be determined by examining the effects of the original variables on the principal components. It can be seen that the proportions of chlorobiphenyls 101 and 118 affect pc-1 positively, whereas the concentration of chlorobiphenyl 153 and the 'sum' have a negative effect on pc-1. Similarly, chloropbiphenyls 138 and 180 affect pc-2 in opposite directions are independent of the other chlorobiphenyls (Fig. 7 in **Paper V**). The 'sum' is the main factor affecting pc-3 (Fig. 8 in **Paper V**). Thus, for example, seal number 8 is an 'outlier' because of the very high concentration of all the chlorobiphenyls (Fig. 5 in **Paper V**). In comparison with harbour or common seals (*Phoca vitulina*), there are relatively few data on chlorobiphenyls in grey seals (*Halichoerus grypus*) (Roots, 1996; 1999).

Juvenile seals are distinguished from adult animals due to the lower sum of the chlorobiphenyl concentrations and the lower relative concentration of chlorobiphenyl 180. There does not appear to be a difference between male and female adult seals. Unfortunately, the sampling set contains only a small number of males. Geographic areas also do not affect the chlorobiphenyl pattern. The sum of the chlorobiphenyl concentrations has decreased considerably since the 1980s, but the relative concentrations have not changed considerably since then. Interestingly, chlorobiphenyl 118 was not detected in the adult females of that year, but this may be an artefact of the analytical technique.

Heavy metals in biota in coastal waters

Heavy metals can reach the marine environment via the atmosphere or through discharges and natural runoff. As a direct impact, annual emissions of heavy metals from the Baltic Sea countries decreased during the period 1996–2000, by 26% for cadmium, 15% for mercury and 10% for lead (HELCOM, 2003). According to estimates, about 9 tonnes of cadmium was deposited in the Baltic Sea during the year 2000. Concentrations of cadmium, lead and zinc are on average higher in the south-western parts of the Baltic Sea, where atmospheric deposition of heavy metals is greater and waste containing high levels of heavy metals has been dumped. One fifth of the cadmium input to the Baltic Sea comes from atmospheric deposition, carried by the prevailing south-westerly winds (HELCOM, 2004a). The atmospheric emissions of cadmium of Estonian power plants have declined from 1 t to 0.7 t in 1997–2000. During the period 1994–2000, discharges of heavy metals, mostly cadmium and lead, also decreased in most of the neighbouring countries. The riverine loads of cadmium and lead in 2000 amounted to about 36 t and 298 t for the Gulf of Finland (Fig. 4 for Cd in **Paper IV**), and 1.5 t and 12 t for the Gulf of Riga. Among other coastal countries, the Estonian input of cadmium and lead in 2000 was proportionally very low, accordingly 0.5 t and 1.9 t for the Gulf of Finland, and 0.04 t and 0.3 t for the Gulf of Riga (HELCOM, 2004a). Information about unmonitored rivers is not available, which theoretically may increase the load of cadmium and lead entering the Baltic Sea from Estonia.

Monitoring in 1994–2001 does not indicate any differences between the contents of heavy metals in the fish from the Gulf of Riga and those from the Gulf of Finland, nor does it indicate any temporal changes or trends (Table 2 in **Paper IV**). Even though the concentrations of some heavy metals have decreased in many parts of the Baltic Sea, like in the Estonian coastal waters, high concentrations can still be found in certain marine organisms, notably in the Baltic herring. For example, mercury concentrations in herring have remained at roughly the same level since the 1980s, but cadmium concentrations in Baltic herring have increased significantly (HELCOM, 2003). The distribution of cadmium in surface sediments is very uneven, ranging from very low levels (0.22 mg kg^{-1} dw) in the Gulf of Bothnia to the highest levels in the Gotland Basin (7.16 mg kg^{-1} dw), the Farö Deep (6.20 mg kg^{-1} dw), and the western Gotland Deep (4.12 mg kg^{-1} dw). Slightly elevated levels are observed in the eastern Gulf of Finland, although the levels in the sediments remain no higher than 1.6 mg kg^{-1} dw in the Estonian coastal sea (Fig. 5 in **Paper IV**). The mapping illustrates the transportation of cadmium and its entrapment in the areas where the bottom waters are anoxic (HELCOM, 2002). The behaviour of cadmium in water is complex and there is a high level of uncertainty in the prediction of cadmium loads entering the marine environment.

Heavy metals in rivers

Long-term, 1994–2003 annual average variations of heavy metals in relation to discharge, measured at the main monitoring stations, show that maximum heavy metal levels occur during autumn–winter, whereas lower concentrations occur during the months of low flow. In general, the heavy metal concentrations in Estonian rivers are around natural background values. This is explained by geochemical factors and the abundance of sedimentary deposits in the drainage basins of rivers, as well as by minimal anthropogenic loads.

A detailed comparison of data is carried out for 2002 and 2003. The range of concentrations of five metals (Pb, Cu, Ni, Cr and Zn) for eight stations is given for 2002 (Table 5 in **Paper IV**). The concentration of Cu was between 1.0–36.0 µg l^{-1} in the rivers in 2002. In a comparison of recent years, an increasing concentration was found in the northern part of Estonia — Keila river 8–19 µg l^{-1} (140 µg l^{-1} in June 2002), Kunda river 10–33 µg l^{-1} (92 µg l^{-1} in December 2002), Pühajõgi, Purtse and Narva rivers accordingly 12, 18 and 36 µg l^{-1}. Regarding the standards for heavy metals, the listed rivers belong to the poor quality class. In 2003, an elevated concentration was found in Kunda river and Mustajõe river, although the status in the major rivers of Northern Estonia improved. The concentration of Cd decreased in several rivers in 2003 in a comparison with 2002. The concentrations of Pb fluctuated between 0.2 and

1.0 μg l^{-1}, showing that Pb stayed at the natural level, and rivers belong to the good quality class. A higher content of Pb was found in North-Eastern Estonia (Selja River 1.0 μg l^{-1}, Kunda River 1.0–4.0 μg l^{-1}, Purtse River 5.0 μg l^{-1}, Pühajõgi 4.0 μg l^{-1}: moderate class). In an annual comparison, the concentration of Pb has increased. The concentrations of Zn were very low, averaging 2–8 μg l^{-1} up to 15–22 μg l^{-1}, being quite stable in the last years. In 2003, higher levels of mercury were detected in Kunda river, in Selja river and in Mustajõgi, accordingly: 1.60, 1.23 and 1.73 μg l^{-1}.

In order to assess pollution sources in the oil-shale region, a comprehensive targeted survey has been carried out in the north-eastern part of Estonia (Table 3 and 4 in **Paper IV**). The concentrations of cadmium in the river sediments are not exceeded in surveyed area in the north-eastern part of Estonia. The highest Cd concentration (0.48 mg kg^{-1} dw) was detected in sediments of the Purtse river near the mouth of the Kohtla river (Fig. 7 in **Paper IV**). The concentration of cadmium reached one third of the target value (0.33 mg kg^{-1} dw, target value 1 mg kg^{-1} dw) near the mouth of the Pljussa river in Russia and in the Kohtla river near the intake of Viru Keemia Group chemical plant. Also, the concentration of mercury in the river sediments was below target values (0.5 mg kg^{-1} dw) in all sampling sites. The highest mercury concentration (0.43 mg kg^{-1} dw) was found in sediments of the Purtse river near the mouth of the Kohtla river. As heavy metals are present in sediments, aquatic organisms can be exposed to these elements. The sediment sampling proves that the rivers have been affected by discharges of oil-shale industries. As expected, the concentrations of heavy metals were much higher in sediment samples than in water samples. The presence of metals in sediments is related to runoff or deposits of water discharge.

To summarise, concentrations of heavy metals are predominantly low in Estonian rivers. In North-Eastern Estonia, concentrations of Zn, Pb, Cd, Ni, and Cr have tended to increase during recent years. For lead and cadmium, there is no prevailing source in Estonia. The major part of the zinc emissions into the sewerage system stem from surface runoff (roofs, streets). The input of heavy metals (especially chromium and nickel) via coagulants into the treatment plant and sewage sludge has to be considered (Baltic Environmental Forum, 2000). According to EU freshwater standards on quality classes (Council Directive 76/464/EEC; Council Directive 78/659/EEC), Estonian rivers are classified by content of heavy metals as good or moderate. Critically important are standards of chemical analysis and detection levels in the assessment, since concentrations vary. In future, the enrichment factor can serve as an indicator of the degree of heavy metal pollution from anthropogenic input into a river. Research is needed to specify whether metals exist in their carcinogenic form, in order to assess their toxicity and impact on biota.

Development of monitoring network for priority hazardous substances

The implemented Estonian national environmental monitoring programme of hazardous substances, which follows EU and HELCOM recommendations, covers all major problem areas, sites, and aspects on a national scale (Fig. 1 in **Paper IV**). The monitoring network has been extended gradually. Operational monitoring by companies, required by the environmental permit system, complements the national network and gives the opportunity for end-of-pipe discharge analysis and for detailed assessment of trends in water-bodies. The multi-scale representation is critically important in the Tallinn area and in North-Eastern Estonia. In other districts, selective monitoring and inspection could provide data for the assessment. Nevertheless, the statistical power of the present sampling is weak; in particular, the temporal frequency should be increased. If the length of a time-series is fixed, the power for various slopes at a certain between-year variation can be estimated (UNEP 2004). Regarding a relatively low standard deviation among the time-series of contaminants in biota, if the desired sensitivity of the monitoring programme is to be able to detect an annual change of at least 5% per year within a time period of 10 years, the power is decreasingly below 30% for sampling in every second and third year at present standard deviation. Databases and inventories of industrial chemicals and hazardous substances should be developed over the next years in addition to the extensive site surveys in places where traces of hazardous substances have been found. The integration of source-oriented and load-oriented approaches is proposed by HELCOM (2004a), since both are lacking full-scale consistent data coverage.

Due to natural conditions, the Baltic Sea ecosystem can be more vulnerable to anthropogenic chemicals than the marine or freshwater environments in other European regions and might call for more stringent measures to combat pollution by hazardous substances. The HELCOM objective with regard to hazardous substances is to prevent pollution of the Convention Area by continuously reducing discharges, emissions and losses of hazardous substances, aiming at the target of their cessation by the year 2020, with the ultimate aim of achieving concentrations in the environment near background values for naturally occurring substances and close to zero for man-made synthetic substances (HELCOM, 2002). For this policy objective, further guidance with regard to identification of relevant sources of release, prioritisation among sources, identification of appropriate measures to cease these releases, identification of appropriate policy instruments to implement these measures, making a choice among the available instruments, and measures aiming to get the best outcome for the effort taken, is needed.

5. CONCLUSIONS

Selection of modelling strategy and reinforcing monitoring network by GIS

The capabilities of geographical information systems for data analysis are improving rapidly. As the performance of desktop GIS software improves and the amount of available data increases, analysts may lag behind in their abilities to use and examine the information. GIS-based environmental models have an essential role in providing effective and efficient environmental services.

Different spatial models have been applied in this research in order to optimise the monitoring network. In order to get an overview of the pollution levels and environmental status in Estonia, a national monitoring system should be developed, together with monitoring of point sources and their dispersion, which would provide a continuous overview of the impact of pollution on nature and of critical loads. This national monitoring programme should deliver data across landscape types, soils, vegetation, and living communities from pollution sources, end-of-pipe locations, and from rivers and the coastal sea. In the long term, the survey quality could be improved by combining conventional analytical methods, making data available, and by modelling. This dissertation proposes a geographical information system approach for analysing the adequacy of environmental monitoring networks by spatially relating monitoring stations, and for efficient allocation of monitoring networks by describing the point patterns of environmental monitoring in Estonia, assessing neighbourhood, configuration and representation on selected criteria, and by analysing the coherence of networks for landscapes and for hazardous substances.

The debate on methods and approaches continues, depending on the scope and objectives. Do we assume representation by typological classes or do we fill the pattern just with 'hot spots'? How do we achieve total national coverage? While studying spatial relations, the primary factor is the phenomenon itself, as shown in case of landscapes, treated as geocomplexes. Essentially, the selection criteria for monitoring methods and sampling strategies adequately follow spatial and temporal relationships for the subject as well as for the comprehensive, areal-based survey. As proven in this dissertation, upgraded monitoring methods, spatial analysis, and spatial functions of the phenomenon enable an integrated analysis in environmental monitoring via established modelling techniques and mapping.

Modelling draws specific conclusions from a set of general propositions stated at the beginning of this dissertation. Four specific objectives were examined and tested and concluded drawn as follows.

1. A modelling algorithm developed to simulate spatial effects in complex situations, to investigate sensitivity of outputs to variation in the controlling

parameters of the model, which is based on a case study of air pollution in the north-eastern part of Estonia.
2. The point patterns of Estonian monitoring networks were assessed, their neighbourhood, configuration and representation on certain criteria examined for the coherence of the environmental strata in order to allocate and optimise the monitoring network.
3. The landscape monitoring network was characterised according to selection of monitoring concepts, methodologies and topologies. Network for landscape monitoring was explored intensively by neighbourhood statistics to propose and reinforce the option of the acquisition of data for multiple purposes from universally designed monitoring programmes.
4. Monitoring network was proposed for hazardous substances in Estonia based on an assessment of the distribution of priority hazardous substances in Estonia and in coastal areas. The comprehensive summary on distribution of priority hazardous substances, persistent organic pollutants and heavy metals in Estonian waterbodies and in the coastal Baltic Sea confirmed the good and moderate status of waterbodies. In addition, the patterns of polychlorinated biphenyls (PCB) in grey seals (*Halichoerus grypus*) from the Baltic Sea were examined to establish a monitoring programme to consider and to assess trans-boundary impacts for the grey seal population in the Baltic.

The results and consequences of research on environmental monitoring and modelling are presented below in abstracts by these four general conclusions.

Spatial modelling of compounding effect of pollution

The link between industrial output and air pollution is complex. Monitoring and modelling form the major tools with which to assess environmental quality, as stated in **Paper I**. It was found that spatial implications of pollution should be assessed. The analysis on air pollution in North-Eastern Estonia was directed at comparing the relative effectiveness of environmental policies considering the non-spatial and spatial characteristics of pollution sources. Single and combined effects were explored. The effect of source contribution was assessed by equal minimisation of emissions at all sources. In average atmospheric conditions, the spatial pattern of sensitivity is such that the sensitivity decreases against the prevailing wind direction from north-east to south-west. The spatially-balanced choice of emissions control is based on findings of the compounded effect of emission change, which tends to select proportionally smaller sources. Thus, there are several options of pollution abatement for the oil shale region of Estonia. In addition, it indicates the need to optimise the monitoring system. The case study indicates cost-effective and spatially-effective policies, which could be undertaken in a harmonised manner for environmental objectives.

Neighbourhood analysis as a basis for monitoring network design

According to the second specific objective, the neighbourhood of the Estonian monitoring network as a whole and according to topical sets was examined. In the context of a point pattern, first- and second-order variations can be related directly to two distinct classes of pattern measure: density-based measures and distance-based measures. Among density-based measures, kernel density estimation provides alternative solutions to the problem of the sensitivity of any density measurement to the variation in a study area. As an alternative, two distance-based measures were applied, such as nearest neighbour distance and complex Ripley's K function, which encompasses information on all intermediate distances in a pattern. The important point is that some preliminary exploration, description, and analysis was needed in identifying an optimal modelling exercise for a monitoring point pattern.

To conclude, networks are designed according to monitoring methodology that tends to populate the sites geographically, cluster in focal areas, or to extend the set throughout different landscape districts, as proven in **Paper II**. The point pattern of environmental monitoring is described using various statistics. The simplest distance measure applied here is the mean nearest-neighbour distance, which was calculated using the set of distances to the nearest-neighbouring site or feature for all sites in the pattern of the monitoring network. Another distance function, K-function, incorporates more distance information in the point pattern to enhance descriptive power, although interpretations were quite complex.

The set of forest monitoring is the only network imposed on a geometrically regular basis. The representation of Estonian districts is different for meteorological, air, water, landscape, biodiversity and soil monitoring. The monitoring sets aiming to acquire data on human impacts are clustered in metropolitan areas. Water monitoring sets are clustered around the lower reaches of river basins. Biodiversity monitoring exhibits strong neighbourhood features in nature reserves. Spatial representation is enhanced through aggregation in square kilometres database, UTM grids, land cover type, or soil type (**Paper III**). It was found that small sampling sets with fewer than 50 stations show biased and statistically insignificant results. As a result of statistical assessment and neighbourhood analysis, appropriate network density and quality, and efficient station configuration could be proposed in order to monitor the environment in Estonia.

The assessment considered development and expansion of the monitoring network according to implementation of European Union directives and pollution prevention policies. In addition, a hypothesis on whether the data of the national monitoring network is sufficient and cohesive for the Estonian average figures, as an indicator in cross-European reporting, is elaborated. European unified policies and measures impact the design of the monitoring network in Estonia, but not *per se*.

Integrated approach for landscape monitoring

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 narrowly not only from specified landscape monitoring programmes but also from other environmental strata, such as biodiversity, forest, soil and water. A key benefit of the use of an integrated approach is that these are legacy sets of intended surveys, produced with specific purpose by qualified methodologies and expertise.

Paper III assesses the neighbourhood of the Estonian monitoring network as a whole in order to test the availability of data and environmental variables for landscape monitoring from multiple sources. The analysis was constructed so that the distance and proximity methods related to topical sets were 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 conventional, 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 embedded in understandings of spatial pattern can be used to support the monitoring of landscapes.

The adequacy of landscape monitoring according to the spatial relation of the environmental monitoring set was explored by means of landscape districts. This method enables us to make decisions by identifying and interactively packaging comprehensive data structures on the level of landscape districts. The validity and transferability of the method to match different data sources at different sites is discussed. Biodiversity sets can complement landscape monitoring in national parks and protected areas. In general, the representation of topical networks on landscapes and land cover types is rather different. For this reason the application of the transfer functions needs further investigation of modelling strategies on small (field), meso-scale (ecotype) and macro (landscape district) level.

A systematic approach in **Papers II and III** assisted in optimisation of 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 on Saaremaa Island. A strategic approach to select monitoring stations is statistically preferable, because proportional samples of districts, which are relatively smaller for large homogeneous districts, are used. The procedure is also 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. Aggregation by regular grid expands options for raster data analysis.

Critical issues that remain are the categorisation and choice of appropriate spatial units that will allow for an integration of landscape variables 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. In addition, 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 examined further with respect to neighbourhood features. Applicability of modern automated techniques, including remote sensing, desktop GIS and statistical software packages, depends on conceptual maturity and flexibility in data management.

Establishment of monitoring network for hazardous substances

The fourth objective of this thesis, monitoring set for hazardous substances, has been examined and developed. In the course of the survey, the distribution of priority hazardous substances has been assessed and mapped. During the last ten years the state of the Estonian environment has improved with respect to hazardous impacts. The major local pollution sources are related to North-Eastern Estonia, where the Estonian energy and chemical industries based on the oil-shale mining are located. Internationally, elevated levels of hazardous substances are associated with the islands of western Estonia, and with southern Estonia, where the long-range transportation of air pollution from central and western Europe result in elevated concentrations of hazardous substances. Long-range transport of chlororganic compounds from southern sources outside Estonia dominates the pollution loads.

As the objective of this sub-task was to determine the relative significance of different hazardous substances in Estonian rivers and in the coastal sea, it is concluded that the freshwater quality criteria are not exceeded in river stations for all heavy metals studied. Metals enrichment may occur during the low flow periods, as well as during autumn–winter. Sediment sample values reflect the proximity of pollution sources, in particular in the Eastern Tallinn industrial zone and in the backyard of the oil shale industries in North-Eastern Estonia.

However, it is impossible to draw any definite conclusions from the limited changes observed in heavy metal concentrations in seawater or marine organisms. Concentrations of some metals, such as cadmium, are declining in organisms in the Gulf of Finland but increasing in the western Baltic Proper. The clear decline in lead concentrations in herring is observed in most areas.

The concentrations of the analysed toxic chlororganic compounds and heavy metals in the Baltic herring of the Estonian coastal sea remain below the standards established by FAO/WHO for fish (FAO, 2001). The coastal waters and sediments do not appear to pose any threat to human health or aquatic life.

The detailed survey on the distribution of PCBs in grey seals is found in **Paper V**. The profile of polychlorinated biphenyls 101, 118, 138, 153, and 180, and the sum of their concentrations in mg/kg lipid in grey seals (*Halichoerus grypus*) from the Baltic, North-East and Eastern England, and the St. Lawrence Estuary (Canada) were examined by means of Principal Component Analysis. The patterns differ between juveniles and adult animals, but the gender of adults and geography do not appear to play a role. The concentration of chlorobiphenyls in grey seals has decreased, but requires further monitoring in proposed methodology and network. As is typical for other species and for other chemicals, the relationship between the concentration of the chlorobiphenyls and health status (Blomkvist *et al.*, 1992) of the species needs further examination.

According to Helsinki Commission data and based on Estonian national monitoring data, the assessment made within the dissertation indicates that the loads of some hazardous substances have fallen considerably over the past 20–30 years, mostly as a result of tighter controls on point source inputs, such as industrial discharges. There is still too little comprehensive knowledge on the impact of the most widely used chemicals and their cocktail-like combinations on human health and on the environment (HELCOM, 2003).

The implemented Estonian national environmental monitoring programme of hazardous substances, which follows EU and HELCOM recommendations, covers all major focal areas, sites, and aspects on a national scale. Operational monitoring by industry supports the environmental permit system and complements the national network, making possible a detailed assessment of trends in waterbodies. Nevertheless, the statistical power of the present sampling is weak; in particular, the temporal frequency should be increased. Databases and inventories of industrial chemicals and hazardous substances should be developed in the future in addition to the extensive site surveys to determine where sources are found. Further, as proposed by HELCOM, it could be fruitful to integrate source-oriented and load-oriented approaches, since both are lacking full-scale consistent data coverage (HELCOM, 2004a). In the course of future developments in environmental management and the implementation of integrated pollution prevention and control (IPPC), monitoring obligations will be shifted towards industrial companies. The public authorities remain in charge of national surveys, assessments, reporting, and inspection.

Final reflection

The potential of environmental monitoring to yield information on the state of the environment should be further examined with respect to national and European initiatives on the development of environmental monitoring networks. Computer-based systems offer the potential to support planning decisions by means of adjusting input criteria and modelling the consequences. GIS applications are now commonplace in a range of such environmental assessments. Technically, there is potential for the methodology developed here to be applied in the model of a planning and design tool, and the discussion offered a detailed commentary on this, concluding that monitoring mapping, whether at the final map level or in disaggregated form, is an important element of the context for monitoring planning decisions.

Key principles for further development of monitoring network and applied research are the following:

- The coverage of monitoring stations across an entire territory
- Setting specific requirements
- To focus on effect of human pressures
- Hierarchical structures (transnational, case studies)
- Continuous temporal surveying and a feasible monitoring cycle

This may create overlapping and further spatial analysis using overlay, distance and proximity functions. It is far better to consider the coverage of stations across an entire territory than to select sites based solely on individual merit. There is convincing evidence that nearest-neighbourhood analysis is applicable to test the monitoring network and gives valuable insight into the monitoring network of landscapes. Representativeness of landscape types, land cover and plant communities is uneven — to be considered in further development of the set. The sets with a small number of stations run the risk of being biased. To improve the natural coverage, a proportional number of stations in each landscape district should be advocated.

Finally, the dissertation provides complementary methodology and information to analysts and policy makers for designing monitoring programmes with which to understand better the status of the environment. It also assists in the better understanding of the linkages between the causes and impacts of pollution, as was explored in the initial stage of the work. The ethos of the work is based on an interdisciplinary approach. The dissertation contributes to monitoring and evaluating the effectiveness of policies addressing landscape monitoring in particular and concerns of distribution of hazardous substances, thus promoting sustainable development and natural resource management. The full potential of GIS for environmental analysis and modelling needs to be explored and discussed further and the research frontiers need to continue to be pushed toward areas that include hierarchies and scales of modelling, seamless interoperability and adaptive web-based solutions.

This research set out to develop a methodology with the potential to support a range of environmental themes and activities, with air pollution, landscape monitoring and monitoring hazardous substances foremost. The approach meets requirements that have been set for environmental monitoring and mapping. It has clearly been shown the importance of methodology, since environmental monitoring is significant in this unspoilt part of Europe and although it currently merits mention in a wide variety of significant policies and reports, unless the experimental aspects of the environment are considered alongside more easily quantified and indicated characteristics, environmental quality can only be partially safeguarded in the future.

6. SUMMARY IN ESTONIAN

Keskkonnaseire võrgustiku optimeerimine integreeritud modelleerimisstateegia ja geoinfosüsteemi abil Eesti näitel

Doktoritöös esitatakse geoinfosüsteemidele (GIS) tuginev modelleerimisstrateegia keskkonnaseire võrgustiku geograafiliseks analüüsiks. GIS-mudeleid rakendatakse seirejaamade ja -võrgustiku paiknemise uuringuteks ning pakutakse välja metodoloogia maastike ja ohtlike ainete seirevõrgustiku optimeerimiseks. Töö eesmärk oli kirjeldada Eesti keskkonnaseirevõrgustiku punktmustrit, hinnata seirejaamade naabrussuhteid, konfiguratsiooni ja teatavate tunnuste esindamist, analüüsida keskkonna teemakihtide ühitamist seirejaamade sidusa ja tõhusa asukohavaliku eesmärgil. Metodoloogia põhineb hajuvusmudelil, punktmustri iseloomustuse ja naabrusanalüüsi statistilistel meetoditel, kartograafia ja ruumianalüüsi meetoditel. Mudeli prototüüp koostati mitmele GIS-tarkvarale, nagu Mapinfo Professional®, Vertical Mapper, Idrisi ja CrimeStat. Mudeli testimiseks ja praktiliseks väljundiks oli Eesti keskkonnaseire süsteem, milles käsitleti põhjalikult kolme uuringut *in casu*, õhusaaste dünaamikat ja muutuste tundlikkust Kirde-Eestis, integreeritud maastikuseire meetodite väljatöötamist ning ohtlike ainete seirevõrgustiku planeerimist tuginevalt ainete geograafilisele levikule. Doktoritöö lähteandmestik pärineb Eesti riiklikust keskkonnaseire programmist, mille üheteistkümne seireteema 68 allprogrammis on kokku ligi 1700 seirejaama või -ala, registreeritavate näitajate arv ulatub 250ni.

Esimeses osas koostati välisõhu saastemudel, mille algoritm rajaneb põhjus-tagajärg seostele tootmisprotsessi, emissiooni ja välisõhu kvaliteedi vahel. Kirde-Eesti õhusaaste mudel lähtub üksiku tööstusettevõtte tasemest, mis võimaldab maakondlike, mitme saasteallika ja saastetasemega juhtumite simulatsioone. Vääveldioksiidi saastetaseme ja selle muutuste kaardid koostati ruutkilomeetrises rastris ja 1980.–1990. aastate kohta. Rasterkaarte analüüsiti ajalises järgnevuses, hinnates statistiliselt muutuste olulisust ruumis ja ajas. Suurimad saasteallikad olid Narva elektrijaamad, mis hõlmasid kaks kolmandikku saastekoormuse ruumilisest muutusest Kirde-Eestis, jättes teiste saasteallikate osatähtsuse selles osas 2–3% vahemikku. Väiksemate saasteallikate mõju on seevastu elastsem, st nende suhteline osakaal muutustes on saastetaseme absoluutarvulisest muutusest olulisem. Kui hinnata muutuste ruumilist tundlikkust piirkondlikult, kahaneb ta alla domineerivat tuult, kirdest edelasse. Samuti hinnati väljundkaartide tundlikkust hajuvusmudeli sisendite suhtes. Testid kinnitavad hajuvusmudeli erksamat reageerimist tuulekiirusele, seejuures on seos pöördvõrdeline. Tundlikkuselt järgnevad mudeli sisenditest emissioon ja tuulesuund. Kokkuvõttes, tööstuse ruumistruktuuri muutuste tulemusena tekivad saastelevi dünaamikas täiesti erilaadsed saasteväljad. Teades välisõhu iga üksiku saasteallika panust piirkondliku välisõhu kvaliteedi muutustes, saab prognoosida saastetaseme ruumilisi muutusi pikemas perspektiivis ning leida optimaalsed lahendused saastekoormuse vähendamiseks.

Seirevõrgustiku asendit ja konfiguratsiooni võrreldi naabrusanalüüsis nii tihedus- kui kaugusnäitajate alusel. Kernel-põhised tihedusarvutused näitavad seirevõrgustiku tihedust teataval alal. Kaugusnäitajana rakendati kaugust lähimnaabrini ja Ripley K-funktsiooni statistikuid. Metsaseire on ainus, millel on geomeetriliselt reeglipärane võrgustik, võrgusilma raadius on 16 km. Maastikuseire võrgustiku ruumiliseks planeerimiseks ühitatakse teemavõrgustikele põhinev andmehange mitmemõõtkavalise põllumajandusmaastikuseirega, et testida naabrussuhete olulisust maastikuseires. Seirevõrgustike paiknemist analüüsiti erineva maakasutuse ja maakatte, elupaikade, mullatüüpide ja inimmõju suhtes, hindamaks integreeritud maastikuseire võimalusi ja seda, kuidas seire teemavõrgustikud täiendavad maastikuandmestikku ning kuidas seirevõrgustik katab maastikurajoone ja eri maastikutunnuseid. Seirevõrgustike 50 km otsinguraadiuses modelleeritud tiheduskaartide liitmisel selgub, et stratifitseeritud, kihilist lähenemist, erinevate seireteemade geograafilist sidumist saab rakendada tulemuslikumalt Tallinna ja Pärnu piirkonnas ning Kirde-Eestis. Selline lähenemine on oluline just komplekssetes loodusuuringutes. Seiresüsteemi edasisel optimeerimisel saab välja joonistada mitu ‘infoauku’. Hõredamalt ja vähetõenäosuslikumalt on seireinformatsiooniga kaetud Raplamaa lõunaosa ning riigipiirialad Pärnumaal ja Viljandimaal. Väiksemad sellised areaalid jäävad Põhja-Kõrvemaale, Pandivere idanõlvale ning Varbla kanti Pärnumaal. Tiheduskaardid on lineaarses sõltuvuses otsinguraadiusest, mis on valitud sõltuvalt seireprogrammi metoodilistest vajadustest, aga ka paiknemismustri iseärasustest. Eesti ruutkilomeetrivõrgustikus on seirejaamadega kaetud vaid 2,7%. Maakatte jaotuses paikneb andmebaasi *CORINE Land Cover* alusel 9% seirejaamadest tehisaladel, 39% poollooduslikel, 43% loodus- ja 6% märgaladel (3% siseveekogudes). Pindalalises suhtes on seirejaamade paigutus kallutatud poollooduslike ja tehisalade suunas. Kõige rohkem, 216 jaama, asub okasmetsades, 168 haritaval maal ja 129 põldudel. Seirejaamade maakattelist esindatust tuleb aga pigem hinnata seireteemade lõikes, mitte kõigi võrgustike summas. Näiteks loomastikuseire on alaesindatud märgaladel, metsaseire oma korrapärase võrgustikuga aga kinnitab enam-vähem proportsionaalset juhuslikku valimit eri metsatüüpide lõikes. Mullatüübiti on üleesindatumad paepealsed ja rähkmullad, mõneti alaesindatud on näivleetunud ja turvasmullad. Kui hinnata maastikulist katvust, siis on seirevõrk nii absoluutskaalas tihedam kui üle keskmise esindatum rannikualadel, Soome lahe ja Liivi lahe rannikumadalikul ning Hiiumaal. Kõrge punktitihedus on Karula kõrgustikul ja ka Palumaal. Hõredamalt katab seirevõrgustik Harju lavamaad ja Lääne-Eesti rannikumadalikku, mis on pindalalt teistest oluliselt suuremad.

Kui iseloomustada seirevõrgustikke teemade lõikes, on näha, et kõige koondunum, agregeeritum on **põhjaveeseire võrgustik** (464 jaama, keskmine kaugus lähima jaamani 1,2 km, tihedaima koondumisala raadius 30 km, hõlmab Tallinna, Kirde-Eesti, Pandivere). Üks koondunuim võrgustik on ka **taime-**

koosluste seire oma (127 jaama). Kuna see võrgustik on koondunum kui kaitsealad ise, annab see kinnitust fokuseeritud looduseuurimisest väiksemal arvul kaitsealadest. Alust oleks ruumiliseks optimeerimiseks, sest taimekoosluste seiret ei tehta intensiivpõllumajanduslikul Järvamaal ja turismikoormusega Otepää kõrgustikul. Kõige hajusam on **meteojaamade võrgustik** (29 jaama), mida geomeetriliste näitajate alusel võib pidada isegi liighajutatuks (keskmine kaugus lähima jaamani 33,3 km). Jaamade arv ja naabrussuhe on selline, et jätab katmata Kõrvemaa, Saaremaa keskosa ja Pärnumaa Läti piiri äärse ala. **Inimmõjutuste jälgimisele suunatud seirevõrgustik** on koondunud linnapiirkondadesse, Põhja- ja Kirde-Eestisse, **elusllooduse seiremuster** on tihedam Lääne-Eestis. Sellise paigutuse üks määrav faktor on kindlasti liikide esinemine ja elupaigad, aga ka looduskaitsesüsteemi ülesehitus ja kaitsealade paiknemine. Statistilised testid näitavad, et **siseveekogude seirevõrgustik** (82 jaama) on koondunud vesikondade olulisemate jõgede alamjooksule, kuid üle-eestiline võrgustik näib ideaalilähedaselt juhuslik. Paarispunktidele üles ehitatud punktmuster avaldub naabrusindeksi jaotuskõveras kõige selgemalt maastikuseires ja siseveekogude seires. Samas peab möönma, et alla 50-jaamalise seirevõrgu puhul ei ole kaugustestid statistiliselt usaldusväärsed. Seirejaamade naabrussuhted on olulised, sest annavad võimaluse interpoleerida näitajaid tuginevalt lähimjaamade andmetele, tuvastada mõjutegurite kaugussõltuvust ja varieeruvust ning leida tunnuspindadele informatsiooni siirdetegurit.

Doktoriväitekirja ühe teesi empiirilise tõestusena hinnati ohtlike ainete geograafilist jaotust Eesti siseveekogudes ja rannikumeres ning sellest lähtuvalt seirevõrgustiku kujundamist. Nagu eeldatud, keskendub ohtlike ainete seire probleemsetele, oluliste inimmõjutustega piirkondadele, kus on tekkimas mitmetasandiline seiresüsteem. Viimase kümnendi andmete analüüs näitab Eesti keskkonnaseisundi paranemist ohtlike ainete osas. Eesti lääne- ja lõunaosas domineerib ohtlike ainete allikana kaugülekanne, Põhja- ja Kirde-Eestis on kaalukam kohalike saasteallikate roll. Ohtlike ainete sisaldus on pinnavetes ja põhjasetetes püsinud enamikus Eesti jõgedes väike, nad kuuluvad Euroopa standardite järgi heasse kvaliteediklassi. Madalveeperioodidel võib siiski suureneda metalliühendite sisaldus. Lisaks füüsikalis-keemilistele on seires kasutusel ka bioakumulatsiooni meetodeid, mis on kinnitanud, et Eesti rannikumere kalades on ohtlikke aineid vähe: nende sisaldus jääb allapoole FAO/WHO toiduainetele kehtestatud ohtlike ainete piirnorme. Ka rannikumere vesi ja põhjasetted ei kujuta ohtlike ainete poolest ohtu inimtervisele ega vee-elustikule. Võrdlevalt uuriti polükloreeritud bifenüülide (PCB) esinemist Läänemere, Inglismaa idaranniku ja Kanadas paikneva St. Lawrence'i estuaari hallhüljestes (*Halichoerus grypus*). Ohtlike ainete levik sõltus hüljeste vanuselisest jaotusest, kuid hüljeste sugu ja täpsem geograafiline paiknemine, lesilad ei mänginud ohtlike ainete esinemise puhul olulist rolli. Ohtlike ainete seire tõhustamiseks pakutakse saasteallikate ja -koormuste seire meetodite integreerimist, kuna mõlemal seiresuunal on andmevajakuid. Samuti tunnistatakse teadmatust kokteililaadsete kemikaaliühendite mõju kohta elusorganismidele. Saasteennetuse meetmete (IPPC) rakendamisel lasub põhiline seirekohustus ohtlike ainete käitlejatel, riiklikud ametkonnad vastutavad üleriiklike inventuuride, hindamiste ja järelevalve eest.

Kokkuvõttes kinnitatakse hüpoteesi, et seirevõrgustiku ruumitunnustel on oluline tähendus seireinformatsiooni kvaliteedi parandamisel, ning et eri seire teemavõrgustike sidumine annab lisainformatsiooni nii uuritava nähtuse kui ka sellega kaasnevate tegurite kohta. Seirevõrgu arendamise küsimust võib püstitada mitmeti: kas me eeldame esindatust tüpoloogiliste klasside järgi pindalaliselt või suurendame paiknemismustri tihedust probleemaladel, jättes üldise üle-eestilise katvuse tingimuse teiseseks. Kuigi seirejaamade paiknemise hindamisel ilmnevad küll teatavad vastuolud seireteema metoodiliste aluste ja lähima seirejaama paiknemise ning võrgustiku konfiguratsiooni vahel, toob pakutav metodoloogia seda põhjendatumalt välja seirevõrgustiku aegruumilise optimeerimise lahendused.

Võtmesõnad: keskkonnaseire; keskkonna modelleerimine; hajumismudel; kaugusstatistika; lähimnaabrus; ruumimuster; ruumiline muutlikkus; õhusaaste; maastikuseire; ohtlikud ained; Eesti; Läänemeri.

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ACKNOWLEDGEMENTS

The work resulting in the enclosed papers was partly funded by the Ministry of Environment of Estonia. Grants to write the thesis, attend conferences and to purchase software were received from the Estonian Science Foundation (ETF), International Association of Landscape Ecology (IALE) and Environmental Systems Research Institute (ESRI).

I am grateful to all of you that have supported my work and have helped make this Ph D thesis comes true. In particular, I would like to express my sincere gratitute to:

– **My father and mother**, who put me on the geographic track from my early days and thanks to whom I received my first exposure to earth sciences and nature.
– **Professor Tõnu Oja**, my supervisor, for provoking me to make a good re-start of my PhD studies, for support throughout the study on environmental monitoring, advocating a deductive process in which hypotheses were tested, starting from the original data, theoretical insight to rigorous argument and clarity and, so, for improving the final result.
– **Dr. Ott Roots**, my co-author, for providing me with a door to the jungle of hazardous substances, for an introduction to the scientific world of persistent organic pollutants and for his magnificent contribution to our two joint published papers.
– **Professor Kalev Sepp**, for his dedicated support in the field of landscape monitoring and for his contribution to our joint published paper.
– **Dr. Jüri Roosaare**, for joint journeys into the Geographic Information Systems and for supervision when reaching my first scientific summits.
– **Dr. Erik Puura**, for giving me the opportunity to have a key role in programming and designing environmental monitoring and for joint research adventures in northeastern Estonia.
– **Professor Ülo Mander**, for mentoring, setting the research challenge, encouraging me to write scientific papers and for tutoring to make connections within the environmental sciences.
– **Dr James Petch**, my MPhil supervisor, for his inspiring enthusiasm, superb directing and essential tutoring in my studies in Manchester Metropolitan University.
– **Raivo Aunap**, for opening up the hidden secrets of cartography, and for his perfect editing in the environmental mapping.
– **Erki Saluveer**, for introducing me to the programming languages for the modular modelling environment, interfaces and for exclusive assistance in mapping approach.
– **Tanel Tamm**, for a rewarding co-operation in the field of spatial data mining, mapping and geographic analysis.

- **Dr. Kalle Remm**, for discussions on the neighbourhood approach and for keeping pace in spatial statistics.
- **Dr. Ain Kull**, for assistance in soil sciences and convincing expert opinions on spatial data sets.
- **Ivar Arold**, for a better understanding of Estonian landscapes.
- **Dr. Ivar Puura**, special appreciation to my scientific friend, for unbounded deep discussions about everything on sciences and art, offering many valuable insights.
- **Eva Liivak**, for giving me excellent technical support and allowing smooth time planning for countless tasks.
- **Ergi Prommik**, for his nimble design support in mapping applications.
- **Tens of highly committed professionals** in the community of Estonian environmental monitoring and their contributions to the debate on the development of a monitoring network, who have provided information on their individual research projects in an initiative to facilitate environmental monitoring in Estonia.
- **My splendid colleagues** in the department of research and development of the University of Tartu, for encouragement, support, and acceptance.
- **Lord Carlisle**, for improving my English.
- **Ilmar Part**, for forming the thesis into correct English.
- **My family and close friends** for joy, true feelings and fun.

The end is the beginning. Thank you all, indeed.

PUBLICATIONS

Spatial analysis of industrial impacts on air pollution: an Estonian case

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Abstract

As Estonia has experiencing rapid structural reform, the contraction has meant that air emissions have decreased dramatically in 1990s. Changing industrial structure may result in a completely different type of air pollution map and a different type of behaviour pattern of pollution load images. Air pollution has compounding effect depending on the temporal and geographical patterns of load, on dispersion, and on deposition, depending on the relative sensitivity of receiving sites. Under these circumstances the substitution between sources can be reached manipulating primary with location factors. The model establishes cause-effect links between production, emissions, and ambient air quality from the case of single-plant to complex situations. The study explores empirical methods to build predictive models of SO_2 emission changes. The spatial analytical capabilities of raster system using approximate diffusion models for SO_2 pollution load in a 1 km grid are explored. The series of images are examined as a whole and changes are explained according their significance and power. Surrogates demonstrate spatial preferences for air pollution control policies in the framework of European Directives.

1 Introduction

Estonia is experiencing rapid changes in industrial structure and in pollution load from industrial activity. Industrial contraction, as the primary factor, has meant that emissions of SO_2 have decreased by 53% between 1990 and 1999. However, Estonian oil shale based power plants are still reported to be among the biggest point source polluters in Europe. From an environmental perspective, the important question becomes deciding how effective abatement policies are in achieving real environmental improvements. Spatial analysis provides a good way of assessing effectiveness of policies.

Here, the idea is to develop a methodology to assess changes of environmental quality in terms of simulated spatial behaviour of polluting sources. There is no strong indication yet that a spatial approach can form the basis of abatement policy practices. This could be a consequence of the practical implementability of spatial instruments rather than understanding of spatial relationship.

2 Methodology

2.1 Modelling strategy

The study aims to provide a modelling methodology for the assessment of the spatial importance of polluting sources, addressing air pollution loads and compounding effects and thereby to provide a means of assessing pollution policies and controlling instrument. A number of specific objectives have been set:

1. To develop a modelling algorithm to simulate spatial effects in complex situations.
2. To investigate sensitivity of outputs to variation in controlling parameters of the model.
3. To examine the spatial importance of effects and to assess management strategies for pollution control.

The spatial relationships of production to pollution output and to load are complex. Sophisticated spatial models and complete data coverage are achievable only in raster data structures, but with conventional environmental data these systems are too coarse for spatial diffusion models which represent continuous fields [6]. A further problem relates to changes over time. Implementing temporal GIS involves simple characteristics changing over time as well as geographical entities changing [4]. If differences on a time axis can be measured, an important issue becomes distinguishing artifacts and errors of measurements and modelling 'true' change from normal geographical variability.

The modelling strategy includes dispersion modelling and spatial module. A sensitivity assessment of model parameters is undertaken to evaluate model structures and behaviour of parameters in different conditions. Finally, the utility of the model for assessing of environmental policies is examined.

2.2 Compounding effect of air pollution

Air pollution has compounding effect depending on the temporal and geographical patterns of load, on dispersion, and on deposition. The distribution of air pollutants depends on the number of sources of different size, on the spatial distribution of sources and on the diffusion process [3]. Their compounding effect has spatial and temporal dimensions. The spatial aspect is analysed in terms of scale, of pattern (clusters or scatters), of configuration (point, linear, areal) and of human induced activities (number, type, magnitude). Changes accumulate over time creating compounding effect, which exceed the simple sum of changes. There is an incremental effect of changes, called by Odum [5] the 'tyranny of small decisions'. Theoretically we can distinguish two approaches of compounded effects. The first is just simple summing up of impacts,

$$\sum_{1}^{n} X = X_1 + X_2 + \ldots + X_n \tag{1}$$

Another approach is expressing compounding processes with probability functions, which simulate the complex behaviour. In practice, the compounding effect of air pollution loads is the simple cumulative effect of a number of sources, which can be considered as a set of cases. In our modelling exercise, 'a lot of polluters of the same pollutant' examines several polluters and one pollutant - SO_2. The analysis is based on spatial dominance of a set of sources at different points in the area.

2.3 Dispersion Model

The model used here is based on the Gaussian plume diffusion model, estimating sulphur dioxide concentrations along an axis. The analytical expression below represents a particular simplified solution of the full-scale diffusion equation. The trajectories of polluting compounds are calculated from the wind field at a level of the atmospheric boundary layer. Both long-term average and actual meteorological conditions are applied. Wind conditions are established using wind rose data. According to meteorological statistics the exponent of the wind speed profile in stability category D is 0.2. The model assumes that meteorological conditions remain constant during travel time, which is adapted for extended periods of release on annual bases. Vertical profiles of wind and specific wind trajectories are excluded. Additionally, if emission rate is large compared with the horizontal diffusion parameters, crosswind variations and Eddy diffusivity may be ignored [1]. Sensitivity assessment of model parameters confirmed the reasonableness of these assumptions about model structure.

These assumptions provide the basis for a long-term, source-oriented model. Dispersion in neutral atmospheric stability conditions and the growth of a plume are described by dispersion parameters determined by wind speed and stack height in the following equation:

$$C_{ij}\,(r,z) = 2\,Q\,/\,r\,\alpha\,u_{sj}\,A_j \tag{2}$$

where C_{ij} (r,z) is the concentration in a sector, i, for particular meteorological conditions, j, (g/m^3); Q is the emission rate (g/s); r is the distance from the source (m); α is the angular width of a sector; u_{sj} is the wind speed at the effective heights of source (m/s) and A_j is the boundary layer depth (m). The concentrations are valid for greater distances where the vertical dispersion coefficient is less than the boundary layer depth A_j. Details about dispersion calculations are provided in Clarke [1].

2.4 Spatial analysis

The results of dispersion modelling are imported to the spatial modelling module. Pollution load at neighbourhood cells is a function of the location of the cell in relation to pollution effects of surrounding sources. The final result of concentrations of pollutants is achieved by integrating, through simple addition, the results of the diffusion model across the entire study area by sources. The OVERLAY module sums up sources in various atmospheric conditions. In time series analysis, two techniques, pairwise and multiple, are used to compare quantitative data images of air pollution. The outcome is the trends in change and the description of characteristic values and the abstraction of anomalies. The function of reclassification is used to divide the distribution into three classes, 2σ, greater than + 2σ and between them, aiming to present 'true change'.

2.5 Inventory of sulphur dioxide emissions

The study area covers the north-eastern part of Estonia, where the industrial sectors are mainly based on oil shale. From the set of 75 point sources 9 large sources, which dominate air quality with 96% of regional SO_2 emissions, are selected for modelling. In the temporal elements of the models, SO_2 emission data for 6 reference years during 1980s and 1990s (1980, 1983, 1990, 1993, 1996 and 1999) are included. An emphasis was laid on updated data coverage of the 1990s, on the annual basis, characterized by the introduction of new air pollution standards, by structural change and by technological shift in the Estonian economy.

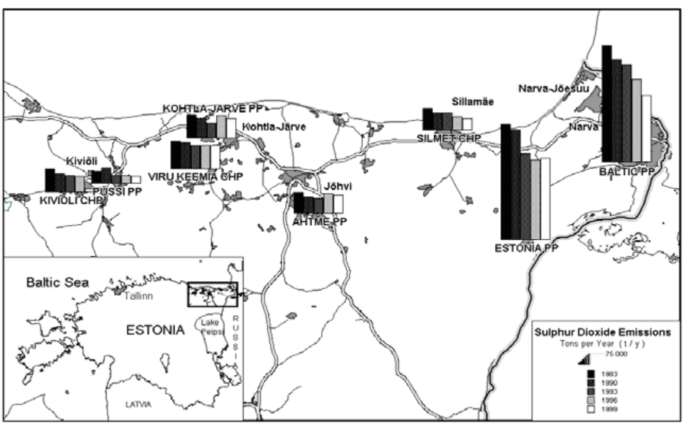

Figure 1: Map of the Large Sources of Sulphur Dioxide Emissions in the Estonian Oil Shale Basin 1983-1999; PP - power plant; CHP – chemical plant.

As we can see from Fig. 1, SO_2 concentrations have been decreased gradually since 1983. To keep track of historical trends, 1980 represents the highest level of combustion of oil shale in the power plants, i.e. historically biggest quantum of emissions, up to 240 000 tons of SO_2 annually. All sources show a clear decline during 1983-90 due to a slow decline in the demand for electricity. In 1990s, the transition markets and decreased demand have been responsible for a substantial decrease in SO_2 concentrations. Gross industrial emissions decreased by 24% from 1990 (150 000 tons) to 1993 (115 000 tons) without substantial environmental measures. Emissions of SO_2 continued to decrease in oil shale power plants by 14% 1993-1996 and by 17% 1996-1999, dominantly due to the decline in the demand for energy. Secondly, this is a results of several technological measures adopted. Improvements in combustion technology enforced by EU requirements are introduced from 1997. From 1999 net electric power output, accordingly emissions, has been steadily increased.

3 Model performance

3.1 Predicted change

The average concentration fields of the modelled SO_2 are given for reference years in the form of isoline raster maps. 99 and 95 percentiles are used to identify changes and to separate them from the errors inherent in dispersion modelling. The thresholds are taken as 3 or 2 standard deviations from the mean. Beyond these limits we interpret true change. Table 1 gives the threshold data, which shows significance change at the 95% levels.

Table 1: Thresholding and areal impact of significance change.

Statistics	Max	Min	μ	STD	Lower limit		Kappa Index of Agreement
Period	μg/m^3				μg/m^3	km^2	
1983-1990	0.71	-17.35	-0.93	1.25	-2.18	705	0.239
1990-1993	3.53	-25.83	-1.57	2.77	-7.11	150	0.218
1993-1996	7.30	-15.17	0.21	2.55	-4.89	56	0.688
1996-1999	-0.11	-8.77	-0.87	0.93	-2.73	189	0.239

Max – maximum change in SO_2 concentration; Min – minimum change in SO_2 concentration; μ – mean change; STD – standard deviation.

Proportionally, the biggest change occurred between 1990 and 1993 (-1.57 μg/m^3). In turn, the change was weakest between 1993 and 1996 (0.21 μg/m^3). Variability of change (STD) characterizes the period of the most dramatic changes from 1990 until 1996, respectively 2.77 and 2.55. In general, areas experience significant decline of SO_2 in an area up to 705 km^2 around Narva in 1983-1990, having less impact in 1990s (up to 189 km^2 1996-1999). The Kappa Index of Agreement performs as the measure of difference between images of SO_2 concentrations. The Kappa index is largest for 1993-1996 changes, 0.688, when the largest area is affected, indicating at changing meteorological conditions. Changes in terms of spatial and population impact are summarized in Figure 2.

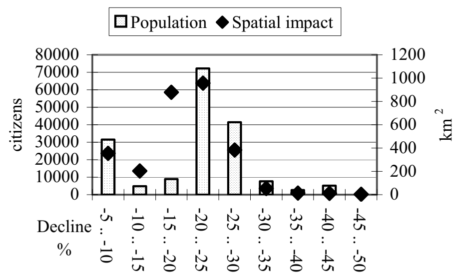

Figure 2: The impact of SO_2 change, 1990-1999.

Over the region there is a clear decrease in SO_2 concentrations, in average between 5-35% in 1990-1999. The most areas belong to the decline 15 to 25%. Instead, the populated areas receive bigger decrease in SO_2 concentrations, at the range 20 to 30%. In particular, a large reduction is predicted by the model in the western part of the district, and between core area and eastern part. Also, changes in SO_2 concentrations over the region are given annually. For instance, the map of relative changes in SO_2 concentrations from 1990 to 1991 is polarized (Fig. 3). The reason for changes is the decline of emissions, but it is complemented by changes in meteorological conditions. The 1990s are exceptional in terms of climate and the inter-annual meteorological variability has to be tracked.

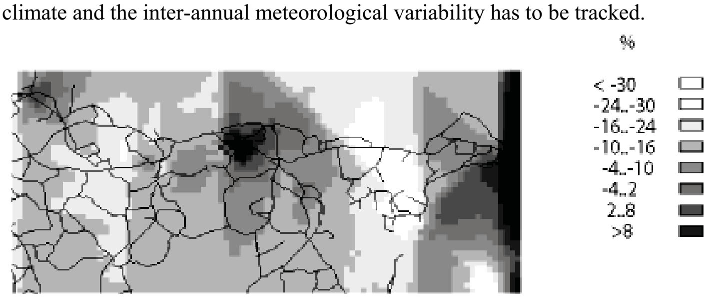

Figure 3: Change in SO_2 concentrations 1990-1991, %.

3.2 Model validation and sensitivity analysis

The test of results of modelling is done using the national monitoring network data in period 1990-1999. This data is obtained from measurements of SO_2 concentrations at four sites twice per day. All four sites are situated in towns. The validation of results is also done comparing predicted concentrations with monitored data for the period 1991-1997 with available on-line measurements. Overall the relation between model response and observations is consistent. The model overpredicts observed values for about 20% of the mean observed value. Large overestimation is confined to areas close to the large sources so the diffusion around the stack is considered as modelling error. Agreement is better around smaller sources. In rural areas, the agreement between modelled and observed values is the best, which redirects the attention at traffic pollution.

The value of the model is not to be judged solely in terms of predicted values. Equally important is the behaviour of the model output in relation to changes in values of the parameters and of the input data i.e. its sensitivity [7]. A small change in some parameters at one site may have a great effect on pollution fields, whereas the pollution may be relatively insensitive to changes in other parameters. In each sensitivity test, one parameter is altered. The combined effect of changing two or more parameters at a time is also examined.

The following analysis is concerned with a change of emission rate and of wind conditions. An analysis of the compounding effect on a regional scale disregards the influence of local circumstances and the sophisticated dispersion rules used in some models. Listed below are the input parameters altered for each test:

- wind direction - change of occurrence by 1% in 45° sector;
- wind speed - change of wind speed by 1%;
- emission rate - change of emission rate by 1%.

The sensitivity results are summarized in Table 2. The test shows that the model is most sensitive to the wind speed, having an inversely proportional relationship. On the other hand, the sensitivity to wind direction change is lowest, +0.08%. Sensitivity assessment of emission rates was based one pollution source, whose impact is described by a particular spatial impact function. In receptors, all sources with their damage functions were added iteratively, the mean change in modelled values is 0.8%.

Table 2: SO_2 concentration sensitivity tests.

Test / Parameter	Mean change %	Mean modelled (1)	1-2 / 2 %
BASIC RUN (2)	-	5.50	-
Wind speed	-1.00	4.95	-10.0
Emission rate	+0.09	5.45	+0.91
Wind direction	+0.08	5.46	+0.72

The indicators of the spatial impact function for 1990-1999 in absolute, relative terms and in regional total are presented in Table 3. The spatial function is assigned to the policy analysis as surrogates describe the importance of pollution load changes.

Table 3: Indicators of change in SO_2 distribution.

Index	A	B	C	D	E	F	G
Source	Emissions change 1990-99		Relative spatial impact		Gross spatial impact		Elasticity
	Ind %	Rel %	Ind	Rel %	Ind	Rel %	F/B
Baltic PP	-53	39.3	0.390	36.0	-20.67	42.0	1.07
Estonia PP	-40	33.3	0.350	32.3	-14.0	28.4	0.85
Viru Keemia	-17	1.2	0.085	7.8	-1.45	2.9	2.4
Kivioli	-87	3.3	0.041	3.8	-3.57	7.2	2.18
Sillamae	-46	1.5	0.024	2.2	-1.10	2.2	1.47
Kunda Cem	-95	7.5	0.027	2.5	-2.57	5.2	0.69
Püssi PP	-71	7.7	0.014	1.3	-0.99	2.0	0.26

A - change of emissions 1990-99, % ; B - regional weight of emission decline, %; C - rate of spatial contribution of regional SO_2 concentration estimated by impact of changed emissions; D - weighted importance of rates of spatial contributions, %; E - actual contribution to the change in regional SO_2 concentrations 1990-99, A*C, %; F - weight of actual spatial contribution of pollution load, %; G - elasticity, ratio of spatial contribution and emission change; Abbr: Ind – individual value; rel –regional, relative; PP - power plant; Cem - cement plant.

It is seen that by all measures the most influential sources are the large power plants. The gross impact is biggest for the Baltic power plant (20%), leaving the Estonian plant second (14%). The relative spatial impact of other sources is 4 to 8 times lower. In average, smaller sources control only around 2-3% of regionwide changes in pollution load. The gross impact of Kivioli is two times bigger than the relative impact, which condition is termed elastic. According to the table, the spatial assessment is setting relevant policy priorities to strengthen spatial impact of controls.

4 Air pollution control strategy

4.1 The impact of EU accession

An air pollution control strategy must refer to the master plan adopted by the national government. In general terms, the strategy is to decrease emissions step by step to achieve EU-standards in the long-term.

Estonia began accession negotiations with the European Union in 1998, the environmental negotiation chapter has been provisionally closed in June 2001. According to agreements, Estonia has undertaken to comply with all the EU directives. In this case, the EU Council Directive 88/609/EEC of 24 November 1988 sets the requirements on the limitation of power plant emission levels, to be applied only to new plants. The old boilers of power plants do not qualify after 2003, before the deadline new technology and precipitations should be installed.

Table 4: Requirements on power plant emissions.

Requirements	Emission limits mg/m^3
EU Council Directive 88/609/EEC of 24.111988 for new power plants, and Estonian regulation on power plants (valid from 2000)	400
EU Commission amendment proposal 98/C 300/04 31.8.98 for new power plants	200
Estonian requirements on existing oil shale power plants (valid from 2003)	2000
Current emission level in Estonian oil shale power plants	1700 - 3000

The Ambient Air Quality Assessment and Management Directive 96/62/EC establishes a framework under which the EU agrees limit values for pollutants [2]. The first Daughter Directive sets limits for sulphur dioxide, nitrogen dioxide, particles and lead to be achieved by January 2005 and 2010. Modelled sources of this study are listed among industrial units of the Integrated Pollution Prevention and Control. Authorities are to be specifying the agglomeration in the north-eastern part of Estonia at which limit values need to be set. The modelling exercise assists in this task defining spatial impact of sources and filtering pollution levels which are exceeded.

4.2 Spatial implications

This analysis is directed at comparing the relative effectiveness of environmental policies taking into consideration the non-spatial and spatial characteristics of pollution source. The single and combined effects are explored. Spatial behaviour of pollution fields for emission changes the following interactive simulations were run:

- what is the effect of a single source;
- what is the double effect of sources;
- what level of emission reduction is required from which source if pollution loads have to fall by 10, 20, ..., n % or by a given amount.

The effect of source contribution is assessed by equal minimization of emissions in all sources. In the average atmospheric conditions, the spatial pattern of sensitivity is shaped so that the sensitivity decreases moving from northeast to southwest, against the prevailing wind direction. The spatial-balanced choice of emissions control is based on findings of the compounded effect of emission change, which tends to select proportionally smaller sources.

4.3 Conclusions

The link between industrial output and air pollution is complex. Modelling forms the major tools to assess air quality. In addition, it indicates the need to optimise the monitoring system.

There are several policy options for Estonia. Oil shale stays the basic primary energy source for next 20 years at least. Keeping wider structural reform ensures the effective use of spatial approach in emission reductions. To reduce by the year 2010 the emission of pollutants to the limits that have been set by the EU requires implementation of a new technology and optimisation of operations for reduced environmental pollution. There is clear indication that cost-effective and spatial-effective policies can be undertaken in harmonized way for the environmental objectives.

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Optimal Set for Monitoring the Environment in Estonia – Neighbourhood Analysis

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Abstract

The paper proposes a geographical information systems approach for analysing the adequacy of environmental monitoring sets by spatially relating monitoring stations. The aim of the paper is to describe the point patterns of environmental monitoring networks in Estonia, assess their neighbourhood, configuration and representation on certain criteria, to analyse the overlapping and coherence of the theme networks and to allocate efficiently monitoring sets. The methodology is based on the statistical description of point patterns and neighbourhood analysis, specifically the nearest neighbour index and Ripley's K-function. A prototype system based on the data model is implemented in Mapinfo, Vertical Mapper and CrimeStat, also the MS Office. The prototype is illustrated and tested by a case study based on Estonia's environmental monitoring system and the spatial presentation of it. The proximity to certain land uses, land cover and human impacts is investigated. Spatial factors results may have implications for the design of monitoring network assuming the national coverage. Evaluation of monitoring sites with mismatch to methodological or least-cost criteria can suggest the optimisation of the set.
Keywords: environmental monitoring; distance statistics, nearest neighbourhood analysis; spatial variation, sampling strategy.

1 Introduction

The paper highlights the inherent difficulties of dealing with conceptual incompatibilities in the pattern analysis, environmental 'reality' and information demand. The objective of the paper is to describe the point patterns of Estonian monitoring networks, assess their neighbourhood, configuration, arrangement and representation on certain criteria, to analyse the overlapping and coherence

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of the theme networks, and to allocate efficiently monitoring sets, to optimise the monitoring network.

Designing the monitoring programme is normally based on the methodology of a singular study or theme, which sets research standards and represents geographical features of a certain natural phenomenon. It has to take into account also the spatial and temporal variability. The comprehensive environmental analysis is impeded as monitoring sets do not overlap in most cases and in a relatively small sampling the chosen set may be subjective due to some practical matters. One has to get an overview of the environmental state of a location chosen at random or in neighbourhood. The importance of the neighbourhood relationships between the monitoring stations lies in the option to interpolate indicators based on the data of nearest stations and in searching for a gradient or transfer function for the area, e.g. meteorological data. On the other hand, discrete phenomena, types of soil or land cover for example, need categorical analysis. The density of sets could be weighed according to the distance zones and neighbourhood. Integration of different monitoring environments and methodologies in different scales and sampling frequencies could create knowledge surplus and the substantial increase of efficiency in the information management, avoiding overload and the enormous quantities of low quality data. The importance of the neighbourhood relationships between the monitoring stations, the purpose of which is to achieve national coverage lies in the following:

- The option to interpolate indicators based on the data of nearest stations
- The distance dependence of impact factors
- Searching for a gradient / transfer function of the study area and identifying the spatial variability

For the background, the Estonian national environmental monitoring set of 11 monitoring themes and of 68 programmes covers almost 1,800 monitoring stations, the number of parameters reaching 250. The temporal frequency of monitoring varies from continuous on-line up to five-year-intervals depending on the dynamics of the phenomenon.

2 Methods

Several statistical techniques can be used for the distribution models. Methods to analyse incomplete data focus on either ignoring the missing values or substituting them with plausible values. A common statement is that a sampling strategy needs to be based on gradients that are believed to exercise major control over distribution of phenomenon. The main environmental gradients can be identified in a preliminary exploratory analysis [1].

As for related studies in Estonia, Kadaja [5] has studied the land cover of Estonia with meteorological information, Meiner *et al* [7] have surveyed the completeness of soil information.

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2.1 Neighbourhood analysis

An approach of density field combines research into a geographical fixed point and its surrounding area. Object- and field-based models can coexist [2]. The methodology is based on the statistical description of point patterns and neighbourhood analysis. The main features of the study are distance parameters or distance statistics. Taking into consideration heterogeneity of study area, the surfaces are described by functions. The density of sets is weighed according to the distance zones and neighbourhood. Definitely, the average density is not the only and the best characteristic for description of monitoring sets. For this reason, monitoring sets are described by distance methods, assessing distances between points or distance from geographical feature. The pattern of monitoring could be modelled on the basis of landscape regions, river basins, land cover and soil typology. MS Office, Mapinfo Vertical Mapper ja CrimeStat were exploited for analysis of the spatial model and data. The thematic maps were produced using Mapinfo. The data source is the national environmental monitoring programme.

One of the oldest distance statistics is the nearest neighbour index. It compares the distances between the nearest points and distances that would be expected on the basis of chance [6]. The formula is simple to understand and to calculate. The distance to the nearest neighbour is calculated and averaged over all points.

$$d(NN) = \frac{\sum_{i=1}^{N} \frac{\mathrm{Min}(d_{ij})}{N}}{\frac{1}{2} \cdot \sqrt{\frac{A}{N}}}, \tag{1}$$

where N is the number of points in the distribution, $\mathrm{Min}(d_{ij})$ is the distance between each point and its nearest neighbour and A is the area of the region. If the observed average distance is about 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 an indicator of first-order spatial randomness. Eventually, the K-order nearest neighbour indices need to be calculated and explored for the investigation of sets and in the comparison. For example, to what extent and in what arrangement surface waters and groundwater monitoring network or plant communities and landscape monitoring networks are related, we could define as proximate locations that influence the given features. K's nearest neighbour distance for each order is defined by:

$$d_K(NN) = \frac{\sum_{i=1}^{N} \frac{\mathrm{Min}(d_{ij})}{N}}{K} \tag{2}$$

The expected nearest neighbour distance is given in the formula:

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$$d(ran)=\frac{K(2K)!}{\left(2^K K!\right)^2 \cdot \sqrt{\frac{N}{A}}}, \quad (3)$$

where K is the order and ! is the factorial operation.

The second function, which is implemented for a model, exercises the Ripley's K-function. It is the 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 [6]. In fact, the Ripley's K-function is the index of non-randomness. As the second order statistic it shows how local clustering is opposed to a general pattern of the set over the region. However, it is also subject to first-order effects so 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:

$$K(d_s) = \frac{A}{N^2}\sum_i \sum_j I(d_{ij}), \quad (4)$$

$$d_s = \frac{R}{100}, \quad (5)$$

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:

$$L(d_s) = \sqrt{\frac{K(d_s)}{\pi}} - d_s \quad (6)$$

3 Results

3.1 Distribution of monitoring stations

First, the distribution of environmental monitoring stations is assessed as a whole. All 1800 national monitoring stations are distributed by river basins and landscape districts, fig. 1. On the basis of area it could be summarised that large landscape districts are proportionally less represented. However, small districts are monitored more intensively, fig. 2. The density of set is substantially higher in the Koiva basin (29 stations). The Western Islands basin is under-represented. In absolute values, the coverage by landscape types is more dense in the coastal plains (Finnish Gulf, Gulf of Livonia; Hiiumaa island). Point density is rather high in the Karula upland and Palumaa.

According to the CORINE land cover [7], 9% of monitoring stations are located in the build-up areas, 39% in the semi-natural areas, 43% in the natural areas (excluding wetlands), 6% in the wetlands and 3% in lakes and rivers, which in general mirrors the distribution of land cover. The highest figure, 216 stations are situated in the coniferous forests, 168 stations in land principally occupied by agriculture and 129 stations in the cultivated fields.

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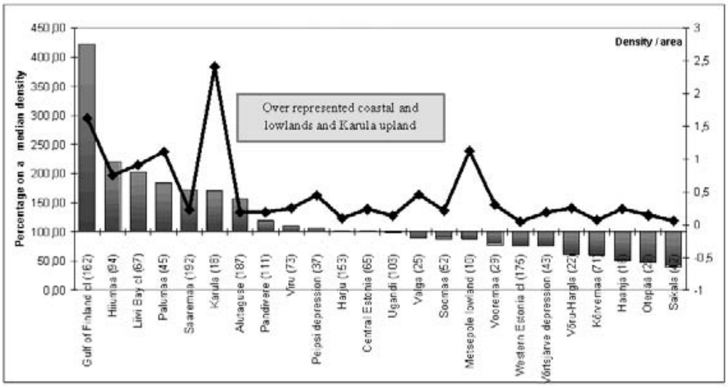

Figure 1. Distribution and frequency of monitoring stations in landscape districts.

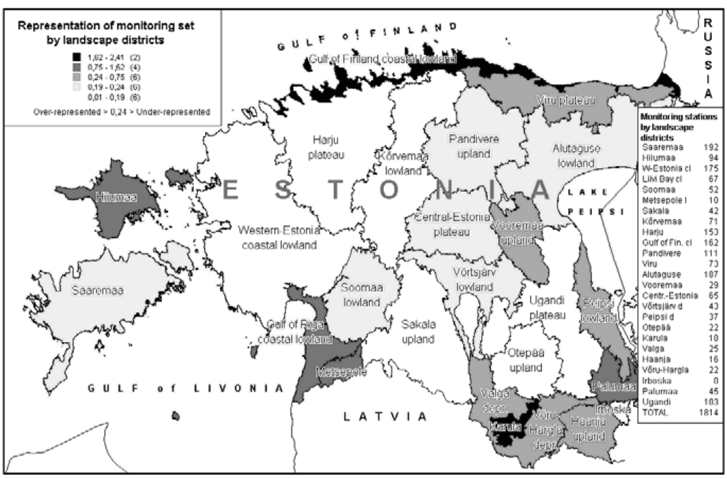

Figure 2. Relative density of monitoring set by landscape districts.

In essence, the density is higher in the build-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 rather

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by topical sets not as whole. Also, this need deeper ecological knowledge and additional data for evaluation. As a result, faunistic monitoring is not widespread in wetlands. Monitoring set of plant groups covers intensely alvars in coastal lowlands. Natural grasslands are over-represented proportionally due to targeted monitoring of rare and endangered species.

The set of forest monitoring confirms more or less proportional random selection through different forest types. Fifty one stations are situated in the coniferous forest, twenty seven stations in mixed forest and nine broad-leaved forest. Soil types, rendsic lepsols and skeletic regosols are over-represented. On the other hand, stagnic luvisols and dystric histosols are under-represented. The main advantage of the listed assessment is that even if the geographic data or some characteristics that the user is looking for are not available, the knowledge from other areas and identified patterns leads to a compromise between needs and availability. The exploration of districts having the same land cover or soil types improves data mining techniques and suits the available sampling set for the goal best. Catogorised data by geographical attributes proves our ability to exploit common object- and field-based analysis functions.

3.2 Neighbourhood statistics of the monitoring set

The examples presented in the previous section demonstrate that categorical analysis is informative having potential value in decision support. Here nine topical sets are assessed by nearest neighbourhood indices. Figure 3 shows clustering of sets. In general, pollution related sets tend to cluster around 'hot spots' with few reference areas represented. The groundwater monitoring set is the most clustered (index 0.18; the average distance to the nearest station is 1.2 km). The set forms clusters in the north-eastern Estonia, in Pandivere, a nitrate sensitive area and in Tallinn metropolitan district, all areas with a significant human impact. The monitoring sets of plant species and fauna are naturally clustered in the protected areas. Compared with other sets the meteorological monitoring is the most dispersed, according to the eucleidic measures even over-dispersed (the average distance to the nearest station is 33.3 km; random nearest neighbour distance is 25.8 km). In addition, the landscape monitoring shows higher dispersion. Consequently, regular, dispersed, aggregated and random patterns of monitoring set are verified.

The curve of k-order nearest neighbour indices shows internal clustering of point patterns of the monitoring sets, fig. 4. The set of groundwater monitoring is clustered up to and including 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. Considering distinction between clustering and dispersion, the set of inland waters monitoring is clustered up to tenth nearest station, which can be defined as river basin area. In general, after the fourth nearest neighbour (rank 4) the differences between sets become less pronounced.

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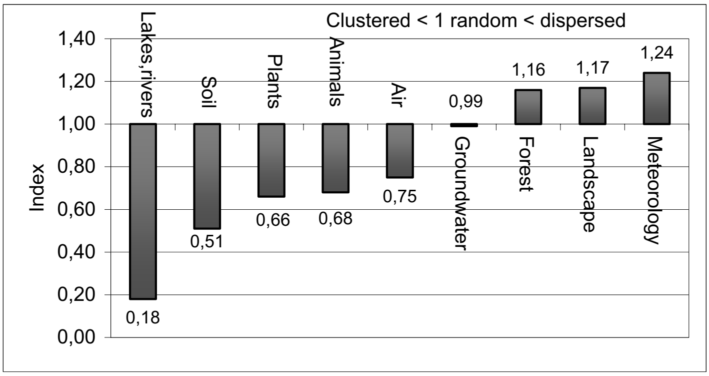

Figure 3. Nearest neighbour index of the Estonian environmental monitoring set.

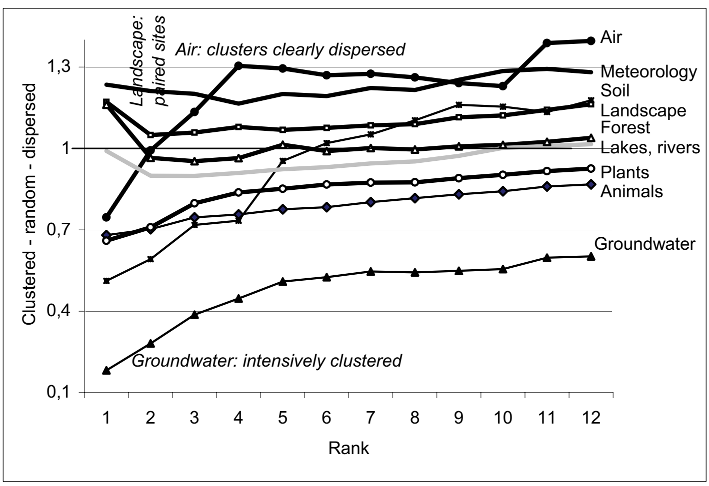

Figure 4. Nearest neighbour index of the Estonian environmental monitoring set by rank.

The complete randomness is not a very accurate foundation so it is possible to compare the distribution of L between the sets and for various baseline landscape characteristics. The Ripley's K-function describes hierarchy of clustering. Clustering is expressed sharply in the set of groundwater monitoring, having a

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radius as 30 km for groundwater bodies. Hierarchical clusters are clearly described in the set of plant species monitoring, where density of point pattern increases up to a search radius of 25 km what represents the size of bigger protected areas. 80km-buffer expresses the distance between nature protection areas. For smaller sets like meteorology and soils, the curve shows the increase of clustering for long distances. According to the Ripley's function monitoring of fauna and forest sets are random and dispersed over longer distances.

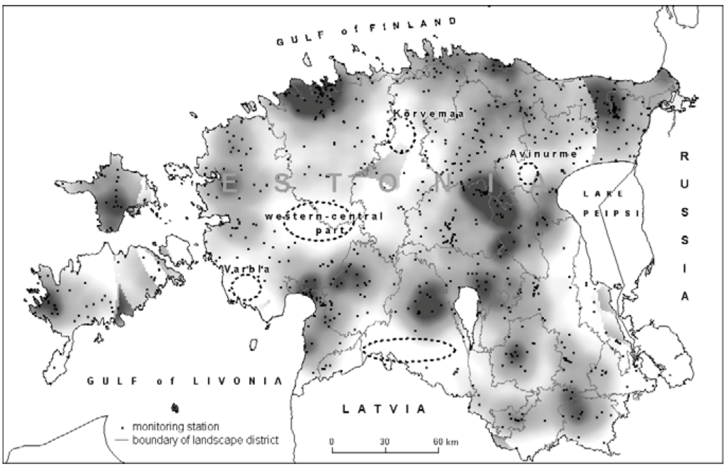

Figure 5. Total density of monitoring sets (50 km search radius).

4 Discussion

4.1 Optimising the monitoring set

Designing monitoring networks to be spatially more feasible is one of the keys of upgrading monitoring methods and decision support systems. The issue is not just to establish new monitoring stations, but to place stations where are no monitors. Figure 5 shows the total density map of monitoring sets of different monitoring themes (50 km search radius). Areas that are more sparsely and uncertainly covered with monitoring information are the western-central part of Estonia and the border areas with Latvia. Smaller "uncovered" areas are depicted in northern Kõrvemaa, in Avinurme and around Varbla. As a density map has the linear dependence on the width of the search buffer, the methodology of the monitoring and the spatial function should be considered. Regarding the 50 km search radius it was assumed that transfer function could be applied for such a distance. Also, 50 km could be taken as the average maximum distance between the monitoring stations.

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First, the monitoring network will be expanded due to implementation of the European Union directives. In particular, the water monitoring needs to be revised in line with the Water Framework Directive. Second, there is currently debate on whether the data of the national monitoring network is sufficient and cohesive for the Estonian average figures, as an indicator in the cross-European reporting. This may affect designing the monitoring network.

Optimising the monitoring set, different location-based models have been applied. The discussion on methods and approaches continues. Do we assume representation by the typological classes or do we fill the pattern in 'hot spots' or do we achieve total national coverage? While studying the spatial relations, the primary factor is the phenomenon itself. The criteria of selection for monitoring methods and sampling strategy follows adequately spatial relationships for the subject as well as for the comprehensive purposes. Therefore, upgraded monitoring methods, spatial analysis methods, behaviour and spatial functions of the phenomenon share applications. Also, the remote sensing could assist in enabling an integrated analysis in applied environmental studies.

5 Conclusions

The article has attempted to assess the neighbourhood of Estonia's monitoring network as a whole and by topical sets. The set of forest monitoring is the only network imposed on a geometrically regular basis. Other sets are designed according to monitoring methodology tending to populate the sites geographically or to extend the set throughout all landscape regions. The representation on landcapes and land cover is different by sets. Small sampling sets having stations below 50 show biased, and it happens, that tests are not statistically significant. Appropriate data density and quality, efficient station configuration and sampling strategies are proposed for monitoring the environment in Estonia.

Acknowlegdements

I wish to thank Tõnu Oja and Kalle Remm for their useful comments and Lord Carlisle for reading through this paper.

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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 (3 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 feed-back 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; Hertzog 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 Hertzog, 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 (Mücher et al., 2000, Griffiths and Mather, 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, 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; Groom, 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 (Fjellstad et al., 2001; Blom and Ihse, 2001). However, there is 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 former 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 co-ordination 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 at 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).

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 (Haase et al., 1986). 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 Hertzog, 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 1,316 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 (Haining, 2003; Upton and Fingleton, 1985; 1989). 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 Office, 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.

$$d(NN)=\frac{\sum_{i=1}^{N}\frac{Min(d_{ij})}{N}}{\frac{1}{2}\cdot\sqrt{\frac{A}{N}}}, \qquad (1)$$

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:

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

$$d_s=\frac{R}{100}, \qquad (3)$$

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:

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

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 multidisciplinary 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 1,200 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, 1980; Teräs, 1985). Counts are carried out in all test areas, to be compared in pairs. Each locality included two transects (2×1,000 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 20×2 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 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. 3).

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. Because 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 (Fig. 4). 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 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 x 16 km. Estonia has 90 monitoring stations with 2 136 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 5), 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 K-function describes the hierarchy of clustering (Fig. 5). 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. Figure 6 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 function 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. 7). 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 3–5 (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 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, 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 Hertzog, 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 Bolm, 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 microorganisms and the number of colony forming microorganisms 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.

Acknowledgements

The work was partly supported by the Estonian Ministry of Education, SRT 0181788s01, SRT 0411767s01, the support scheme for doctoral students and the Estonian Science Foundation, grant No. 6007. The research project was also supported by the Estonian Environmental Monitoring Programme and by the Ministry of Environment of Estonia. We thank Ilmar Part for language editing.

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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 (gamma) diversity index; contagion index.

Table 2. Environmental monitoring programmes complementary to landscape monitoring in Estonia

Pprogramme	No. Station	Main parameters	Frequency	Working scale	Size of test site	Representation and modelling techniques
Agricultural Landscapes	18	Landscape structural elements, bioindicators	Every 5th year	1:5 000 1:10 000	2 x 2 km	Categorical mapping, landscape typologies, transects
Coastal landscapes	26	Landscape elements, plant diversity	Every 5th year	1: 5 000	1x 0.5 km	Categorical mapping, typology of coastal habitats; historical mapping
Remote Sensing of Landscapes	7	Landscape cover classes, landscape structural indicators	Every 5th 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 5th year	1:1 000	50 x 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 5th year	1:10 000	100 m transect up to 10 x 10 km	Transect, major nesting sites
Biodiversity: rare plant species	225	Status of populations of rare plant species	Every 5th year	1: 10 000	2x2m; 10 x 10 m	Categorical mapping of land cover
Biodiversity: soil's biota	17	Earthworms, soil micro-organisms	Every 5th 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 x 16 km grid; classification; generalised linear model, spatial autocorrelation; kriging
Soil monitoring	8	Quality of soil	Annually	1: 10 000	50 x 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 x 1 km up to 50 x 50 km	Dispersal model
Meteorological	59	Data on meteorology and hydrology	On line	1:10 000	1 x 1 km	Dispersal model, kriging

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

Land cover classes by CORINE	Total	Stations per 100 km^2	Monitoring network by environmental strata								
			Meteorology	Air	Groundwater	Rivers, lakes	Landscape	Plant com.	Faunistic	Forest	Soil
Continuous urban	2	40.54		1	1						
Discont. 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-irrig. 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				

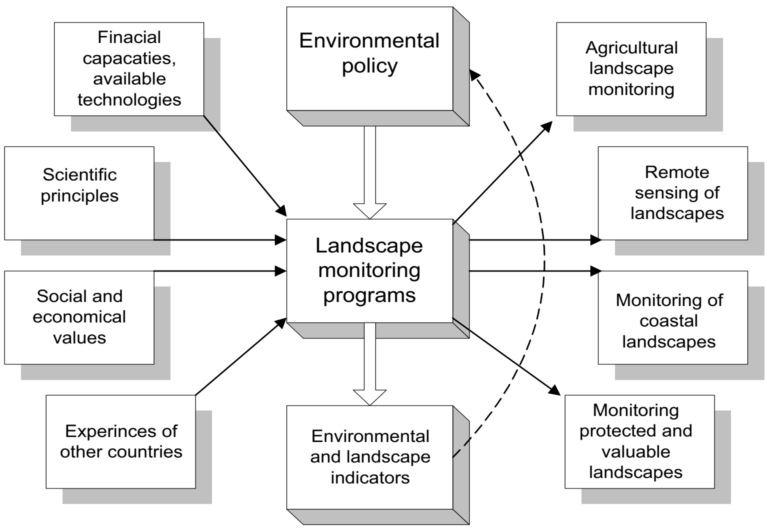

Fig 1. The concept of the landscape monitoring programme in Estonia as applied here

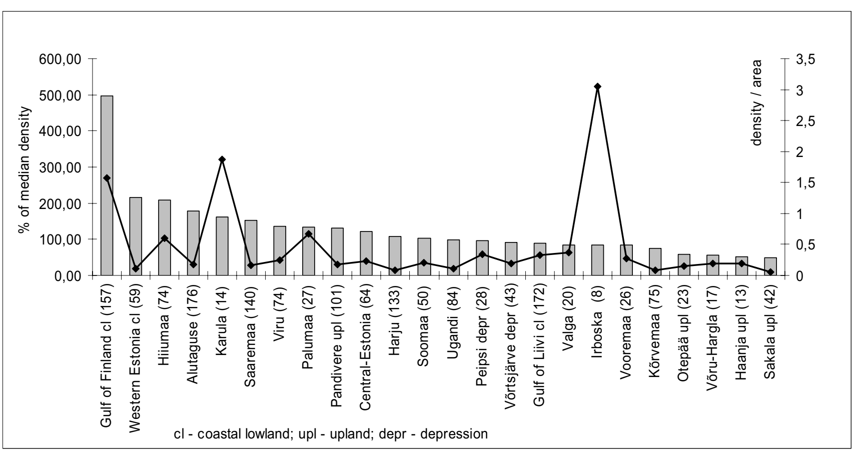

Fig 2. Distribution and frequency of monitoring stations in landscape regions

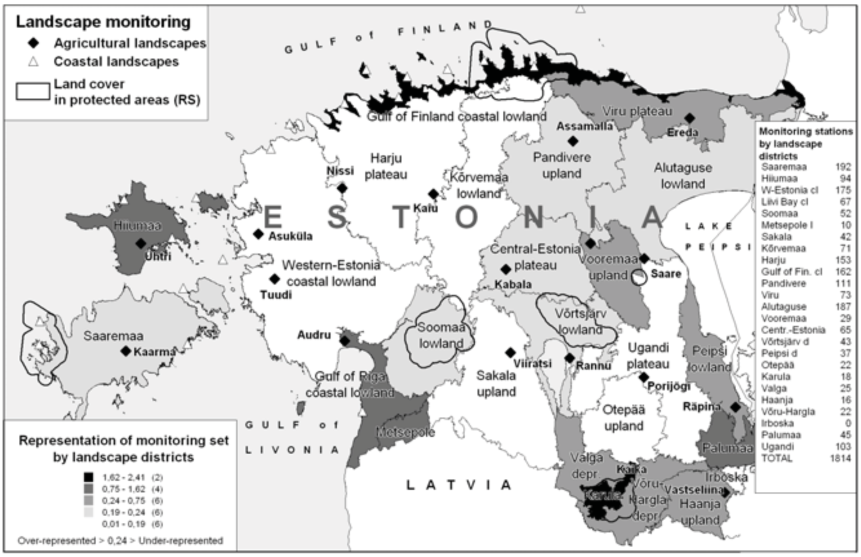

Fig 3. Distribution of landscape monitoring set and density of all sets by landscape regions

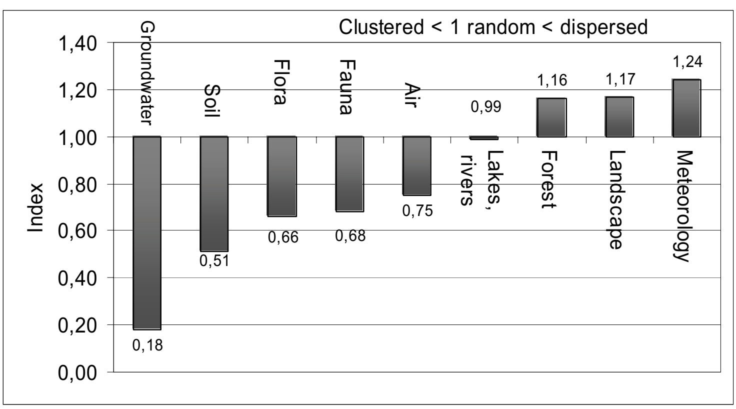

Fig 4. Nearest neighbour index of the Estonian environmental monitoring set

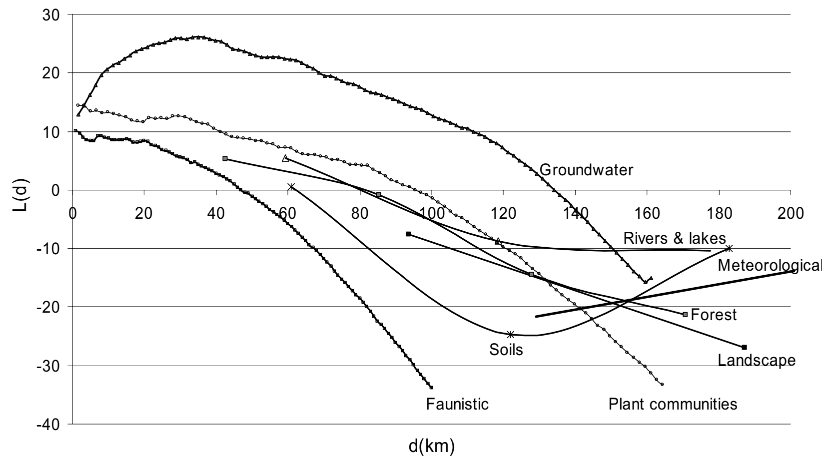

Fig 5. Density of monitoring stations according to Ripley's K-function

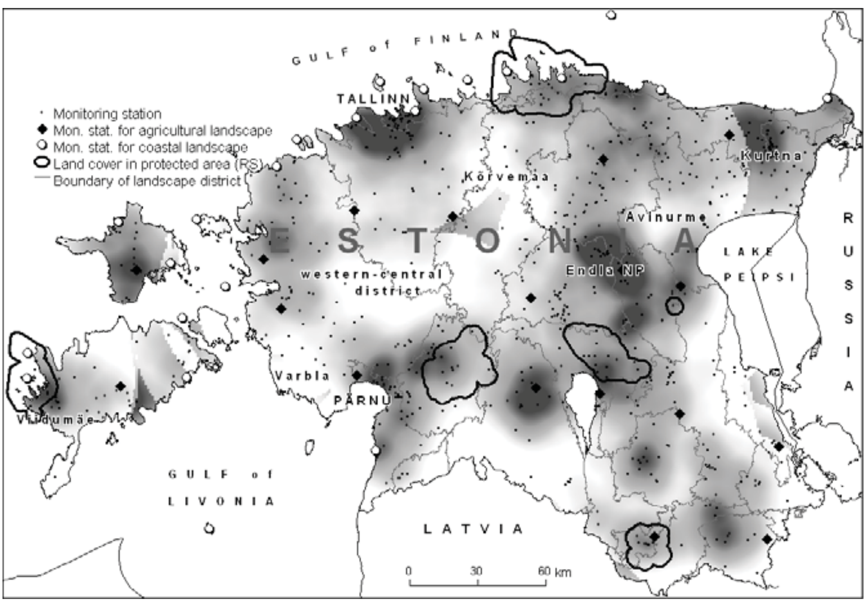

Fig 6. Total density of monitoring sets (50 km search radius, dark - low density, light – high density)

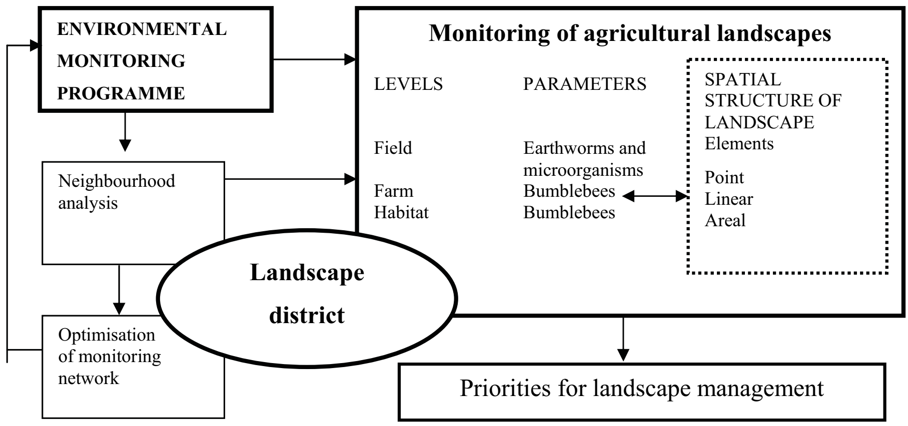

Fig. 7. Framework for applied integrated landscape monitoring

BOREAL ENVIRONMENT RESEARCH 10: 89–102
Helsinki 25 April 2005 ISSN 1239-6095

Monitoring of priority hazardous substances in Estonian water bodies and in the coastal Baltic Sea

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Roose, A. & Roots, O. 2005: Monitoring of priority hazardous substances in Estonian water bodies and in the coastal Baltic Sea. Boreal Env. Res. 10: 89–102.

The water sub-programme of the Estonian National Environmental Monitoring Programme aims to monitor and to develop an information support system for the protection of inland surface waters, transitional waters, coastal waters and groundwater. Focusing on problem areas and reflecting intensive human impact, monitoring of hazardous substances is targeted at populated industrial metropolitan areas in Tallinn and in the oil-shale region of north-eastern Estonia. During the last decade the state of the environment regarding priority hazardous substances has continuously improved. According to monitoring results, the concentration of hazardous substances in sediments and in surface water remains low in the majority of Estonian rivers, and their quality by European standards is classified as good. Concentrations of hazardous substances found in Baltic fish in the Estonian coastal sea remain below standards established by the FAO/WHO for food. The key to the improvement of monitoring is the integration of source-oriented and load-oriented approaches, since both are lacking full-scale consistent data coverage.

Introduction

During the last ten years (1994–2003) the state of the environment regarding priority hazardous substances has continuously improved in Estonia (Roose *et al*. 2003). Although a lot of information is available, a comprehensive overview of priority hazardous substances is not available due to lack of data management, cross-national synthesis, and integrated framework projects in this field in Estonia. Reasons for the present trends: in majority, declining concentrations of hazardous substances, are not fully understood. Also, data and discussion in the context of the Baltic Sea Basin as a reference area is needed.

Efforts have been made to structure and manage national environmental monitoring activities since the early 1990s. As a result, the Estonian National Environmental Monitoring Programme (NEMP) was initiated in 1994 (Roots and Saare 1996). Presently there are altogether around 1800 monitoring stations in the monitoring set of 68 sub-programmes of 11 monitoring themes, the number of parameters reaching 250. Several NEMP projects are related to the European networks or regional projects in the Baltic Sea Basin and are founded on an international framework of standards, methodology and reporting. The following is the current list of important applied programmes in the field of hazardous substances:

— Cooperative Programme for Monitoring and Evaluation of the Long-range Transmission

of Air Pollutants (EMEP);
— Helsinki Commission COMBINE programme;
— UNEP Chemicals Persistent Organic Pollutants Global Monitoring Programme;
— International Cooperative Programmes (ICP) under Geneva Convention: ICP for Assessment and Monitoring of air Pollution Effects on Forest; ICP for Assessment and Monitoring of Acidification of rivers and Lakes; ICP on Integrated Monitoring of Air Pollution Effects on Ecosystems, etc. (UNEP 2002, HELCOM 2004b).

The political objective stated by the EU is to achieve concentrations in the water environment that are near background values for naturally occurring substances and close to zero for man-made synthetic substances (European Parliament 2002). Regarding environmental targets, the objective of the water sub-programme of NEMP is to establish a monitoring and information support system for the protection of inland surface waters, transitional waters, coastal waters and groundwater.

The aim of this paper is to summarise the results of the monitoring of priority hazardous substances, persistent organic pollutants and heavy metals in the Estonian water bodies and in the coastal Baltic Sea. This work is part of efforts to collect and evaluate all available monitoring data on priority hazardous substances in Estonia and in the coastal area. The article focuses on toxic priority substances that are listed in the Stockholm Convention and in UNEP Transboundary Air Pollution Convention protocols of persistent organic pollutants and heavy metals. The protocols require a ban or the minimisation of these priority substances (UNEP 2003). Persistent organic pollutants (POPs) and heavy metals (HM) are a group of toxic and persistent chemicals whose effect on human health and on the environment include dermal toxicity, immunotoxicity, reproductive effects and teratogenicity, endocrine disrupting effects and carcinogenicity (UNEP 2003). Import of chlororganic pesticides to Estonia was prohibited by a government regulation from 1967. Estonia itself has not been manufacturing chlorine organic pesticides.

In the first stage of drafting water management plans, the types of water bodies are determined, their status is assessed, and the water bodies are classified on the basis of existing monitoring data. As the concentration of POPs in rivers was below detection level, the focus of the survey shifted to the fish species of the coastal sea. It is essential for human health that all countries monitor potentially hazardous chemicals in food supplies. Many chemical contaminants are readily taken up by plankton, fish, birds, and mammals and become concentrated at the top of the food chain in marine mammals and fish (Wieder *et al*. 1998, Roots and Zitko 2004). In the human uptake of POPs in Finland, fish and fish products accounted for 82%, and Baltic herring *Clupea harengus* alone for 52% of the total intake (Kiviranta *et al*. 2001). The Estonian data is compared primarily with monitoring data from Finland and Sweden (Ukonmaanaho *et al*. 1998, Heikkilä 1999, Agrell *et al*. 2001, HELCOM 2004b, Voigt 2004). Due to the substantial differences in monitoring programmes the comparison of Estonian data with Latvian and Lithuanian data is in general unfeasible. Despite that surveys of heavy metals allow to draw some conclusions about the status of the Baltic states (Klavins *et al*. 2000, Klavins and Vircavs 2001).

Material and methods

Study area

In Estonia, the contents of toxic chlororganic compounds in the ecosystem of the Baltic Sea have been surveyed since 1974 (Roots 1996). The sampling time and location, i.e the population location, play an important role when different regions are compared. According to the objectives of the hazardous substances monitoring programme, the monitoring stations are predominantly located in the metropolitan area surrounding Tallinn and in the oil-shale region in north-eastern Estonia, reflecting intensive human impact. Monitoring stations of NEMP to survey priority substances are shown on the map in Fig. 1.

The article uses data from four monitoring programmes of priority substances:

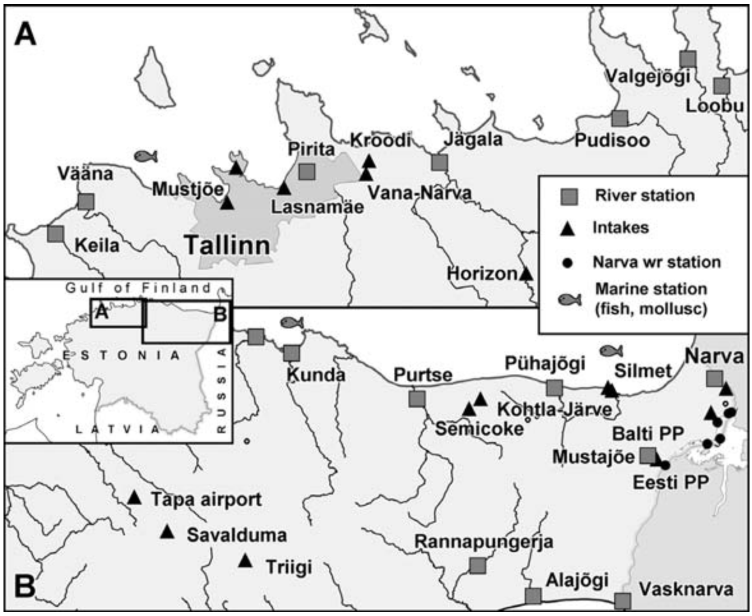

Fig. 1. Map of monitoring set of priority substances of the Estonian National Monitoring Programme (NEMP).

— rivers: sampling fisheries, in particular,
— intakes: sampling sediments as content in the water is extremely low,
— Narva reservoir: sampling water,
— coastal sea: sampling biota.

Overview and details of the monitoring programme of priority substances in the coastal sea, rivers and intakes, including the list of sampled toxic substances, is given by Roose *et al*. (2003). The extensive data set comprises water quality data from the national water registry and environmental monitoring programme. Although data on organic pollution is often incomplete, with some years missing, it is sufficient for identifying spatial and temporal trends in rivers and in marine water composition, and the impacts on living organism.

Chlororganic substances in fish and molluscs are sampled annually in three monitoring clusters (Pärnu Bay, Tallinn and Kunda Bay). Consistent with recommendations of the Helsinki Commission (HELCOM), the selected bio-indicator is the female Baltic herring of two–three years of age (HELCOM 2004b). In the case of zoobenthos, only the content of metals is analyzed in *Macoma baltica* and *Saduria entomon*. Monitoring samples are collected once a year from three to five points in the southern part of the Gulf of Finland. In Estonia, heavy metal content in the ecosystem of the Baltic Sea has been surveyed since 1974 (Jankovski *et al*. 1996), whereby comparable results are from the second half of the 1980s. In fish, the heavy metal content has been determined in their livers. Baltic herring have been caught in the autumn from the northeastern part of the Gulf of Riga as well as from Pärnu Bay and from the two areas (Tallinn and Kunda) in the Gulf of Finland.

Regarding the sampling in rivers, the concentrations of heavy metals are assessed in 15 Estonian rivers. The frequency of sampling of heavy metals is seasonal. Data series for water quality began in 1992 and have continued up to today. In 1999–2001, the inventory reports of hazardous substances in intakes (sediments and water) have been published separately for three Estonian counties: Lääne- and Ida-Virumaa (oil-shale region), and Harjumaa (Tallinn and its surroundings), and jointly for all other Estonian counties.

Taking into consideration the results of inventories, a new programme for monitoring hazardous substances of intakes has been launched in 2002. The design of the monitoring programme originates from the research methodologies and location specifics of a certain natural phenomenon, taking into account also the spatial changeability of the phenomenon. The pilot sampling was split into three stages (Roose *et al.* 2003). Sampling in the first pilot year focused on northeastern Estonia (Fig. 1B), and in the second year, on the metropolitan area (Fig. 1A).

In general, the state of environment in Estonian water bodies depends directly on the efficiency of wastewater treatment and measures applied in industrial processes. The decrease in pollution load in the beginning of the 1990s was caused by a decline in industrial production. In the late 1990s up to the present, the improvement has been achieved with the construction and massive renovation of treatment plants (Kristensen and Hansen 1999, EEA 2003). Also, the amount of wastewater has decreased through the years, due to decreasing water consumption. In general, the water quality in Estonian water bodies, both rivers and lakes, is good or satisfactory (EEA 2003). By the content of organic matter the water quality is good. Main problems are related to phosphorus pollution in some northern rivers (Loigu and Leisk 2002). There are some rivers in the oil-shale region where concentrations of phenols and hydrocarbons are higher than European standards.

Sampling

The Estonian Environmental Research Centre, where all the samples of the coastal sea and intakes programmes were analysed, is acknowledged by the German accreditation bureau Deutshes Akkreditieruns-system Prüfwesen GmbH (DAP) DAP-PL-3131.00 (2008-11-22). Description of sampling techniques as well as the analytical procedures for persistent organic pollutants can be found in Roots (2001). For quantitative determination of polychlorinated biphenyls (PCB) congeners, the internal standard IUPAC 189 was added. PCBs were analysed on a 90 m capillary column (DB-5) using gas chromatography (Varian 3380) with an electron capture detector (ECD). PCB isomers with IUPAC numbers 28, 52, 101, 105, 118, 138, 153 and 180 were analysed. The detection limit for different PCBs was 1 μg kg^{-1} fresh weight.

The analysis of heavy metals follows ISO 8288-1986 (E) (ISO 1994). Analysis uses AAS VARIAN SpectrAA-250 Plus atom absorption spectrophotometer with graphite and flame furnaces. The analysis encompassed the evaluation of heavy metal levels in river and marine environments, to assess potential effects and to identify pollution sources. Detection limits are of great importance in the analysis of river water as concentrations of heavy metals are very low. Higher detection limit may affect much higher load estimates than the actual load. Detection limits of two Estonian laboratories which are involved in sampling are given in Table 1. Long-term metal pollution in the Gulf of Finland has also been documented in bioaccumulation studies of the widespread bottom species, which exhibited elevated concentrations of Hg, Pb, Zn, and Cd (Ukonmaanaho *et al.* 1998, Sipiä *et al.* 2002). Having determined the amounts of heavy metals in rivers, in the liquid phase, and in the sediment, one can calculate heavy metal mobility, bioavailability, and toxicity values (Leivuori 1998, Wieder *et al.* 1998, Toro *et al.* 2001). The methodology of ecological risk assessment is not applied in this article as the scope is purely to introduce monitoring results.

Results and discussion

Persistent organic pollutants in the Estonian coastal waters

The distribution of PCBs in the surface sediment reported in HELCOM (2002) suggests that 'hot

Table 1. Detection limits (μg l^{-1}) for river water in Estonian laboratories.

Laboratory	Cu	Cd	Pb	Zn	Hg
Estonian Environmental Research Centre	1.0	0.02	1.0	10	0.05
Tartu Environmental Research Ltd	0.1	0.02	0.2	2	0.1

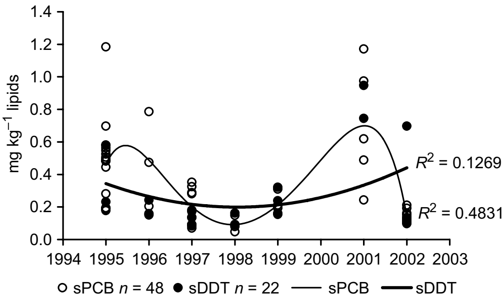

Fig. 2. Contents of sDDT and sPCB (mg kg^{-1} lipids) in the muscle tissue of Baltic herring from different parts of the Estonian coastal sea (Roots and Simm 2003).

Fig. 3. Comparison of dioxin concentration in muscle tissue of the Baltic Sea herring in Estonia and in Finland (Roots et al. 2003).

spots' have been identified in several locations, but not in the Estonian coastal sea. The highest level of PCB contamination was observed in the eastern Gotland Basin and in Lübeck Bay. Higher concentrations were also found near Stockholm, Viipuri and Klaipeda. To summarise, PCBs had entered the environment in large quantities for more than 37 years and were bio-accumulating and depositing in sediments (Koppe and Keys 2001). The role of long-range transport dominates in Estonia and its coastal sea, though the interaction of (airborne) POPs with surface media is not sufficiently understood (Scheringer *et al.* 2004).

The concentrations of DDT and PCBs in the tissues of Baltic herring decreased in 1995–1998, but there was a certain rise after 1998 (Fig. 2). However, the reasons for the increase are unclear (Roots and Simm 2003). It is possible that DDT have recently been used and discharged from Latvian or closely adjoining territory (Olsson *et al.* 1999). In the area of the Baltic Sea during 1994–1998, the highest DDT and PCB concentrations in herring muscle tissue were found near the German coast. The lowest PCB concentrations were found along the Estonian coast, but also in the northern Bothnian Bay and in the Kattegat (Olsson *et al.* 2002b).

European Union (EU) Council Regulation 2375/2001 put the threshold limit value of PCDD/Fs in fish at 4 pgTEQ g^{-1} wet weight. The Baltic Sea fish have been separately highlighted because, in terms of PCDD/Fs content, they may presumably exceed the threshold, in particular for older Baltic Sea herring. The comparison of dioxin concentration in the muscle tissue of the Baltic herring in Estonia and in Finland shows statistically reliable correlation between concentration of dioxins and age of fish ($r > 0.8$) (Fig. 3). The highest dioxin contaminated herring were found in the Gulf of Bothnia (HELCOM 2003). According to the data from HELCOM (2004b) the highest dioxin concentrations for Baltic Sea herring were found in the central Baltic. No differences were found between the commercial landings of Baltic Sea herring in 1996 and 1999 (Karl and Ruoff 2004). The accumulation of organochlorines in salmon might have been increased by their feeding on relatively older specimens of herring and, more especially, on the whole age-range of the more slowly growing sprat (Vuorinen *et al.* 2002).

Concentrations of HCH-isomers (lindane) in water and biota have decreased considerably since the early 1980s. Concentrations of dioxin and PCBs in marine ecosystems declined in the 1980s, but this decrease levelled off in the 1990s. Dioxin levels in fish still exceed the new EU food safety limits in some areas, particularly further north in the Baltic Sea. Concentration levels of POPs are still so high that they have potential biological effects, at least in the Kattegat, the Belt Sea, and the Sound. The conditions differ substantially between the Baltic Proper and the Estonian coastal sea. The dioxin congener profiles in the Estonian coastal sea from herring in the western Gulf of Finland are similar to those from the central Baltic; those from the middle of the Gulf of Finland are similar to those from the Gulf of Riga (Roots and Zitko 2004). Of the twelve Baltic herring samples taken from Estonian coastal waters and the central Baltic, the dioxin content of only one of them (a fish older than 6 years and more than 17 cm in length from the central Baltic) was above the internationally permitted threshold. For other endocrine disrupting substances and new contaminants like flame retardants, a full assessment of their levels or effects is not possible due to the lack of monitoring data.

The concentrations of chlororganic substances in the biota of the Estonian coastal sea do not exceed quality standards set by the EU (Council Regulation 2375/2001). The risk of toxicants is determined by acceptable daily intake (ADI), the exceeding of which could be dangerous to health. The dose is given by weight. In this case, the amount of fish eaten should be taken into consideration. The second risk indicator that is commonly applied is the highest concentration of the substance that does not affect test animals (NOEL, No-Observed-Effect-Level) (Wexler 1998). The latter is applicable for infants and the elderly.

Heavy metals in biota in coastal waters

Heavy metals can reach the marine environment via the atmosphere or through discharges and natural runoff. As a direct impact, annual emissions of heavy metals from the Baltic Sea countries decreased in 1996–2000, by 26% for cadmium, 15% for mercury and 10% for lead (HELCOM 2003). According to estimates, about 9 tonnes of cadmium was deposited in the Baltic Sea during the year 2000. Concentrations of cadmium, lead and zinc are on average higher in the south-western parts of the Baltic Sea, where atmospheric deposition of heavy metals is greater and waste containing high levels of heavy metals has been dumped. One fifth of the cadmium input to the Baltic Sea comes from atmospheric deposition, carried by the prevailing south-westerly winds (HELCOM 2004a). Atmospheric emissions of Estonian power plants have declined from 1 t to 0.7 t in 1997–2000.

In 1994–2000, discharges of heavy metals (mostly cadmium and lead) decreased in most of the sub-regions neighbouring Estonia. The riverine loads of cadmium and lead in 2000

Table 2. Concentrations of cadmium in the biota in the Estonian coastal areas (Simm and Roots 2003).

Organism	Marine area	Period	n	mg kg^{-1} dry weight	mg kg^{-1} wet weight
Saduria (whole organism)					
	Klooga	1990–1995	50	0.72 ± 0.04	0.18 ± 0.01
	Kakumäe	1988–2001	69	0.94 ± 0.04	0.23 ± 0.01
	Käsmu	1988–2001	125	1.00 ± 0.06	0.25 ± 0.01
	Kunda	1988–2001	78	0.86 ± 0.06	0.21 ± 0.01
	Narva	1990–2002	114	0.96 ± 0.06	0.23 ± 0.02
Macoma (soft web)					
	Klooga	1990–2002	40	1.03 ± 0.10	0.22 ± 0.03
	Kakumäe	1988–2002	130	1.49 ± 0.07	0.27 ± 0.01
	Käsmu	1988–2001	116	1.11 ± 0.06	0.22 ± 0.01
	Kunda	1988–2001	99	1.63 ± 0.10	0.28 ± 0.02
	Narva	1990–2002	65	1.65 ± 0.11	0.25 ± 0.02
Baltic herring (muscles)					
	Gulf of Riga	1994–1999	89	0.09 ± 0.03	0.02 ± 0.01
	Baltic Proper	1994–1999	87	0.02 ± 0.00	0.01 ± 0.00
	Tallinn	1994–1999	291	0.11 ± 0.02	0.02 ± 0.01
	Kunda	1994–1999	52	0.27 ± 0.16	0.06 ± 0.04
Baltic herring (liver)					
	Gulf of Riga	1994–2002	34	1.72 ± 0.20	0.44 ± 0.04
	Baltic Proper	2002	1	1.18	0.36
	Tallinn	1994–2002	172	2.47 ± 0.12	0.55 ± 0.03
	Kunda	1994–2002	34	1.50 ± 0.11	0.42 ± 0.03
Perch (liver)					
	Pärnu	2002	10	0.36 ± 0.03	0.09 ± 0.01

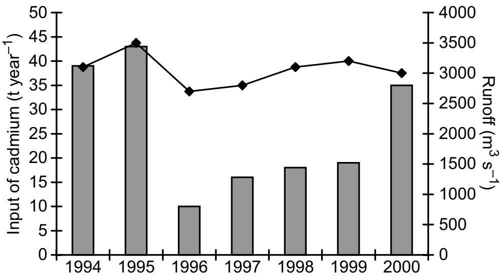

Fig. 4. Riverine input of cadmium (t year^{-1}) and annual average riverine runoff (m^3 s^{-1}) and into the Gulf of Finland, 1994–2000 (HELCOM 2003).

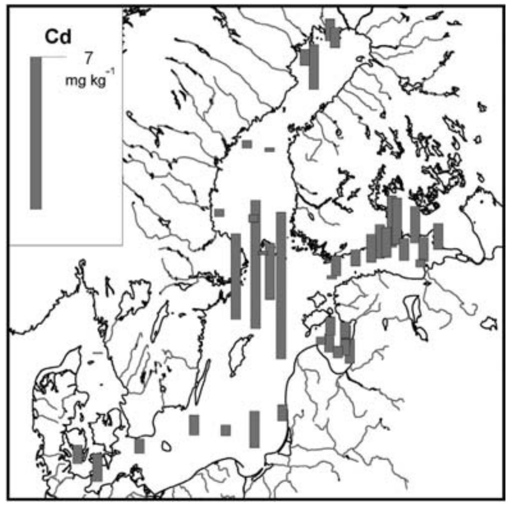

Fig. 5. Distribution of Cd levels in surface sediment in the Baltic Sea (data from HELCOM 2002).

amounted to about 36 t and 298 t for the Gulf of Finland (Fig. 4 for Cd), and 1.5 t and 12 t for the Gulf of Riga. Among other coastal countries, the Estonian input of cadmium and lead in 2000 was proportionally very low, accordingly 0.5 t and 1.9 t for the Gulf of Finland, and 0.04 t and 0.3 t for the Gulf of Riga (HELCOM 2004a). Information about unmonitored rivers, which theoretically may increase the load of cadmium and lead entering the Baltic Sea from Estonia, is not available.

Monitoring in 1994–2001 does not indicate any differences between the contents of heavy metals in the fish from the Gulf of Riga and those from the Gulf of Finland, nor detects any temporal changes, or trends (Table 2). Even though the concentrations of some heavy metals have decreased in many parts of the Baltic Sea, like in the Estonian coastal waters, high concentrations can still be found in certain marine organisms, notably in the Baltic herring. For example, mercury concentrations in herring have remained at roughly the same level since the 1980s, but cadmium concentrations in Baltic herring have increased significantly (HELCOM 2003). The behaviour of cadmium in water is complex and there is a high level of uncertainty in the prediction of cadmium loads entering the marine environment. The lowest Cd concentration in herring liver was found in the Kattegat and the highest concentration in the central part of the Bothnian Sea (Olsson *et al*. 2002a).

The distribution of cadmium in surface sediments was very uneven, ranging from very low levels (0.22 mg kg^{-1} dw) in the Gulf of Bothnia to the top levels in the Gotland Basin (7.16 mg kg^{-1} dw), the Farö Deep (6.20 mg kg^{-1} dw), and the western Gotland Deep (4.12 mg kg^{-1} dw). Slightly elevated levels were observed in the eastern Gulf of Finland although the levels in the sediments stay no higher than 1.6 mg kg^{-1} dw in the Estonian coastal sea (Fig. 5). The mapping illustrates the transportation of cadmium and its entrapment in the areas where the bottom waters are anoxic (HELCOM 2002).

Persistent organic pollutants in rivers

The survey in the target areas near Tallinn and in the oil-shale region indicated (Tables 3 and 4) that concentrations of all tested pesticides of sewage water was below target levels. Also, concentrations of aldine, dieldrine, endrine, DDT, hexachlorocyclohexane and hexachlorobenzene in the sediment samples were below permitted target levels. The concentration of POP was below detection level. In total, the use of agricultural chemicals decreased dramatically in Estonia in the 1990s. On the other hand, monitoring set has been extended gradually by governmental institutions.

Using the total water discharge into the Baltic Sea via rivers (475 km^3 y^{-1}) and using the median concentrations of 0.7 ng l^{-1} PCBs, 0.06 ng l^{-1} of

DDTs and 0.1 ng l^{-1} of HCHs, river transport results in an annual quantity of 332 kg of PCBs, 2.8 kg of DDTs, and 47.5 kg of HCHs to the Baltic Sea. At the beginning of the 1990s, the rivers and the atmosphere contributed about equally to the PCB load in the Baltic Sea, while for pesticides, atmospheric deposition was about 5–7 times more important (Fig. 6) (Agrell *et al*. 2001).

The PCBs hot spot in river water, nearest to the Estonian coast in the Gulf of Finland is situated in the mouth of the Neva river (Russian Federation) and its tributatory. According to the data by Shushkin (1997), the highest concentration of PCBs was in the mouth of Okhta river (1.5 μg l^{-1}). High PCB levels were also registered in the Chernaya river, near the Bely island, and the Sestroretsk coast. Another site of elevated PCB concentrations close to the Estonian coast in the Gulf of Riga is in the mouth of the Daugava river (Nordic Env. Research Programme 1999). Since the water exchange rate at this location is high, it is possible that the discharge may affect the Gulf of Riga and even the Baltic Proper (Olsson 1999, Olsson *et al*. 1999).

Heavy metals in rivers

Long-term (1994–2003) annual average variations of heavy metals in relation to discharge, measured at the main monitoring stations, show that maximum metal levels occur in autumn–winter, whereas lower concentrations occur during the months of low flow. Similarly to Latvia, the metal concentrations in Estonian rivers were around natural background values. This may be explained by geochemical factors and the abundance of sedimentary deposits in the drainage basins of rivers in Latvia, as well as by minimal anthropogenic loads (Klavins *et al*. 2000, Klavins and Vircavs 2001).

The range of concentrations of five metals (Pb, Cu, Ni, Cr and Zn) for eight stations is given for 2002 (Table 5). The concentration of Cu was between 1.0–36.0 μg l^{-1} in the rivers in 2002. In a comparison in recent years, an increasing concentration was found in the northern part of Estonia: Keila river 8–19 μg l^{-1} (140 μg l^{-1} in June 2002), Kunda river 10–33 μg l^{-1} (92 μg l^{-1} in December 2002), Purtse, Pühajõgi and Narva

Table 3. Hazardous substances in the river sediments in north-eastern Estonia, 2002.

Substances	Unit	National target value	Kohtla river: impact of VKG	Purtse river: impact of VKG	Eesti power plant	Narva Vesi: treatment plant	Balti power plant	Pljussa river: mouth
Aldrin	ng g^{-1}		< 5	< 5	< 5	< 5	< 1	< 1
Dieldrin	ng g^{-1}		< 5	< 5	< 5	< 5	< 1	< 1
Endrin	ng g^{-1}		< 5	< 5	< 5	< 5	< 1	< 1
DDT	ng g^{-1}		< 5	< 5	< 5	< 5	< 1	< 1
Lindane	ng g^{-1}		< 5	< 5	< 5	< 5	< 1	< 1
HCB	ng g^{-1}		< 5	< 5	< 5	< 5	< 1	< 1
PCB	ng g^{-1}							< 5
Hg	mg kg^{-1}	0.5	0.03–0.04	0.05–0.43	0.02–0.04	< 0.02	0.045–0.047	0.131–0.132
Cd	mg kg^{-1}	1	0.196–0.331	0.250–0.484	< 0.25	< 0.25	0.119–0.123	0.329–0.331
PAH	mg kg^{-1}	5						0.15
Petroleum hydrocarbons	mg kg^{-1}	100				36.9		
Sn	mg kg^{-1}	10	0.283–0.724	0.255–1.026		< 0.25		
Ni	mg kg^{-1}	50	3.54–3.84	5.14–15.50		1.44		
Cu	mg kg^{-1}	100	6.11–11.70	5.10–17.50		2.8		
Pb	mg kg^{-1}	50	2.5–4.01	10.0–15.8		< 2.5		
Zn	mg kg^{-1}	200	14.1–25.2	19.0–61.4		9.06		
Cr	mg kg^{-1}	100	10.3–11.6	5.56–6.96		< 1.25		
Mono-basic phenols	mg kg^{-1}	1					0.14	
Di-basic phenols	mg kg^{-1}	1					2.0	

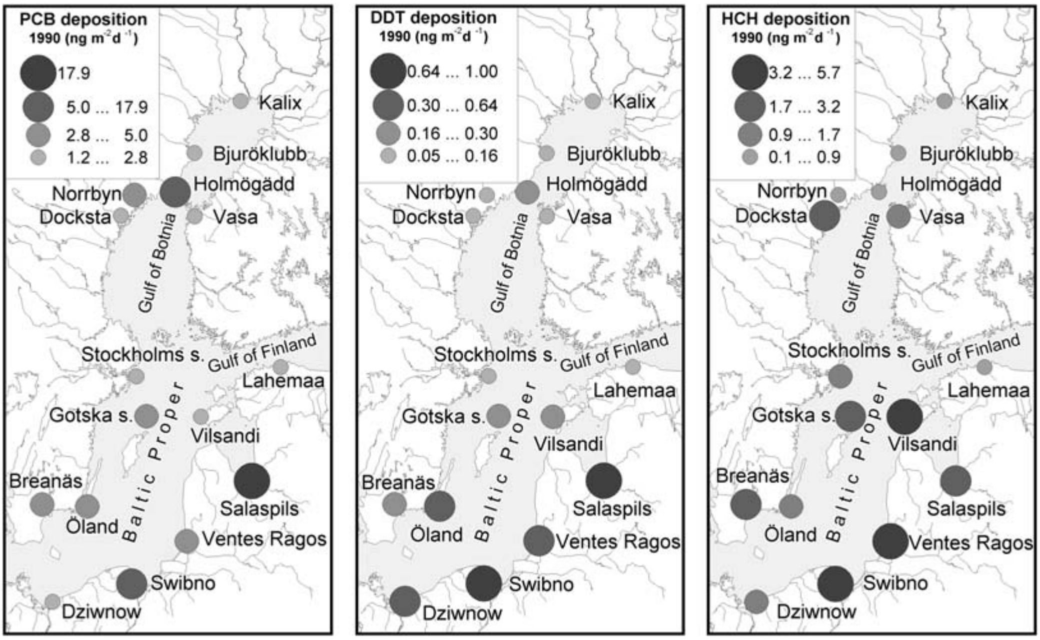

Fig. 6. PCB, DDT and HCH calculated depositions at the stations in the Baltic Sea Basin (data from Agrell et al. 2001).

Table 4. Hazardous substances in the water in north-eastern Estonia, 2002.

Substances	Unit	Kohtla river after VKG discharge	Kohtla river in Lüganuse	Purtse river after Kohtla	Purtse river: mouth	Narva Vesi: treatment plant	Pljussa river: mouth
Aldrin	ng l^{-1}	< 10	< 10	< 10	< 5	< 10	< 10
Dieldrin	ng l^{-1}	< 10	< 10	< 10	< 5	< 10	< 10
Endrin	ng l^{-1}	< 10	< 10	< 10	< 5	< 10	< 10
DDT	ng l^{-1}	< 10	< 10	< 10	< 5	< 10	< 10
Lindane	ng l^{-1}	1	< 10	< 10	< 5	< 10	< 10
HCB	ng l^{-1}	< 10	< 10	< 10	< 5	< 10	< 10
1,2-dichloro-ethane	µg l^{-1}	< 1	< 1	< 1	< 1	< 1	< 1
Chloroform	µg l^{-1}	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1
Trichloro-ethylene	µg l^{-1}	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1
Tetrachloro-ethylene	µg l^{-1}	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1
Carbon tetrachloride	µg l^{-1}	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1
Hg	µg l^{-1}	0.05	< 0.05	< 0.05	< 0.05	< 0.05	< 0.05
Cd	µg l^{-1}	0.1	< 0.1	< 0.1	< 0.1	0.1	< 0.02
PAH	µg l^{-1}						0.011
Petroleum hydrocarbons	µg l^{-1}					87.6	
Sulphide	mg l^{-1}					< 0.02	
Sn	µg l^{-1}					< 0.005	
Ni	mg l^{-1}					< 0.001	
Cu	mg l^{-1}					0.018	
Pb	µg l^{-1}					0.002	
Zn	µg l^{-1}					< 0.01	
Cr	mg l^{-1}					< 0.001	

rivers accordingly 18, 12 ja 36 μg l^{-1}. Regarding the standards for heavy metals, the listed rivers belong to the poor quality class. In 2003, elevated concentrations were found in Kunda and Mustajõe rivers although the status improved in major rivers of northern Estonia. The rise of the cadmium concentration was examined in all rivers. Higher concentrations, between 0.23 and 0.85 μg l^{-1}, were detected in Keila, Pirita, Loobu and Jägala rivers (moderate class). Concentration of Cd in the majority of rivers was in the range 0.02–0.1 μg l^{-1} (good class). The concentration of Cd decreased in several rivers in 2003 as compared with that in 2002. The concentrations of Pb fluctuated between 0.2 and 1.0 μg l^{-1}, showing that Pb stayed at the natural level, and rivers belong to the good-quality class. Higher content of Pb was analysed in north-eastern Estonia (Selja river 1.0 μg l^{-1}, Kunda river 1.0–4.0 μg l^{-1}, Purtse river 5.0 μg l^{-1}, Pühajõgi 4.0 μg l^{-1}: moderate class). In an annual comparison, the concentration of Pb has increased. The concentrations of Zn were very low and quite stable in the last years, averaging 2–8 μg l^{-1} up to 15–22 μg l^{-1} in the samples taken from Keila, Pirita, Jägala and Kunda rivers. In 2002, the concentration of mercury was below analytical detection (0.1 μg l^{-1}). In 2003, higher levels of mercury were detected in Kunda river, in Selja river and in Mustajõgi, 1.60, 1.23 and 1.73 μg l^{-1}, respectively.

To summarise, concentrations of heavy metals were predominantly low in Estonian rivers. In north-eastern Estonia, concentrations of Zn, Pb, Cd, Ni, and Cr have tended to increase during the recent years. This is because of a large quantity of sewage and industrial wastewater flowing into the rivers. According to EU freshwater standards on water quality classes (Council Directive 76/464/EEC, Council Directive 78/659/EEC), the Estonian rivers are classified regarding their content of heavy metals as good and moderate. Critically important are standards of analysis and detection levels in the assessment, since concentrations vary. In future, the enrichment factor can serve as an indicator of the degree of heavy metal pollution from anthropogenic source into a river.

In general, in regions with poor equipment, waste water overflows can be of high importance for heavy metal emissions to surface waters. Measures for source control have to be evaluated in every single case. For lead and cadmium, there is no prevailing source. The major part of the zinc emissions into the sewer system originates from surface runoff (roofs, streets). The input of heavy metals (especially chromium and nickel) via coagulants into the treatment plant and sewage sludge has to be considered (Baltic Environmental Forum 2000). Further analyses are required to specify whether metals exist in their carcinogenic form, in order to assess their toxicity and impact on biota.

In order to assess pollution sources in the oil-shale region a comprehensive survey has been carried out in the north-eastern part of Estonia (Tables 3 and 4). The concentrations of cadmium in the river sediments were not exceeded in the surveyed area in the north-eastern part of Estonia. The highest Cd concentration (0.48 mg kg^{-1} dw) was detected in sediments of the Purtse river near the mouth of the Kohtla river (Fig. 7 and Table 3). The concentration of cadmium reached one third of the target value (0.33 mg kg^{-1} dw, target value 1 mg kg^{-1} dw) near the mouth of the Pljussa river

Table 5. Concentrations of heavy metals (µg l^{-1}) in Estonian rivers in 2002 (Hannus et al. 2003).

River	Cu	Cd	Pb	Zn	Hg
Kasari in Kasari	1.5	0.03	0.2	4	0.10
Keila in mouth	8.0–19.0	0.05–0.54	< 0.2–0.8	3–22	< 0.10
Pirita, Lükati bridge	2.6	0.29	0.2	19	< 0.10
Pärnu, Oreküla	3.0	< 0.10	< 1.0	< 10	< 0.05
Selja, mouth	2.0	0.06	1.0	10	0.15
Kunda, mouth	10.0–33.0	0.06–0.08	1.0–4.0	< 10–21	0.10–0.65
Narva , Narva	36.0	0.09	< 1.0	< 10	0.13
Emajõgi, Tartu	< 1.0–2.5	< 0.02–0.04	0.4–1.0	3–9	< 0.10

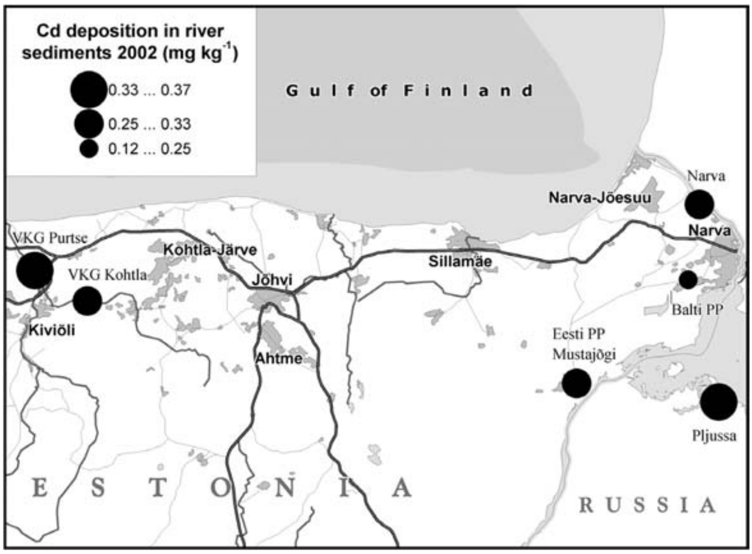

Fig. 7. Cd deposition in river sediments in the north-eastern part of Estonia.

in Russia and in the Kohtla river near the intake of Viru Keemia Group. Also, the concentration of mercury in the river sediments was below target values (0.5 mg kg^{-1} dw) at all sampling sites. The highest mercury concentration (0.43 mg kg^{-1} dw) was found in sediments of the Purtse river near the mouth of the Kohtla river. The sediment sampling proves that the rivers have been affected by discharges of oil-shale industries for decades.

The concentrations of Cd and Hg were lower than target values for intake quality. Pb, Cu, Ni, Cr, Zn were sampled in the Narva river after the outlet of the water treatment plant of the Narva City. The concentration of mercury in the Narva water reservoir was 0.13 mg kg^{-1} dw. The concentrations were above the permissible limits. As heavy metals are present in sediments, aquatic organisms can be exposed to these elements. As expected, the concentrations of heavy metals were much higher in sediment samples than in water samples. The presence of metals in sediments is related to runoff or deposits of water discharge.

Conclusions

During the last ten years the condition of the Estonian environment improved in respect to hazardous impacts. The major pollution sources at the regional level are related to north-eastern Estonia, since the Estonian energy and chemical industries are based on the oil shale mined there. Internationally, elevated levels of hazardous substances are associated with the islands of western and southern Estonia, where the concentrations of hazardous substances are elevated due to long-range transportation of air pollution from central and western Europe. Long-range transport of chlororganic compounds (PCB, etc.) from southern sources outside Estonia dominates in pollution loads.

As the objective of this article was to determine the relative significance of different hazardous substances in the Estonian rivers and in the coastal sea we can conclude that the freshwater quality criteria are not exceeded in river stations for all metals studied. Metals enrichment may occur during the low flow periods, as well as during autumn–winter. Sediment sample values reflect the proximity of heavy-metal sources, in particular in the eastern Tallinn industrial zone.

It is not possible to draw any general conclusions from the limited changes observed in heavy metal concentrations in seawater or marine organisms. Concentrations of some metals, such as cadmium, are declining in organisms in the Gulf of Finland but increasing in the western

Baltic Proper. The clear decline in lead concentrations in herring is observed in most areas. The concentrations of the analysed toxic chlororganic compounds and heavy metals in the Baltic herring of the Estonian coastal sea remain below the standards established by FAO/WHO for fish (FAO 2001). The coastal waters and sediments do not appear to pose any threat to human health and aquatic life. The same is true for riverine and atmospheric inputs of organic contaminants, though not enough accurate data are available to allow detailed analysis.

According to the Helsinki Commission data and based on Estonian national monitoring data, our assessment indicates that the loads of some hazardous substances fell considerably over the past 20–30 years, mostly as a result of tighter controls on point source inputs such as industrial discharges. On the other hand, critical sources should be further monitored and remedial actions taken. There is still too little comprehensive knowledge about the impact of the most widely used chemicals and their cocktail-like combinations on human health and the environment (HELCOM 2003).

In order to get an overview of the levels of air and water pollution in Estonia, a monitoring system should be developed across the country, simultaneously with monitoring of local air and water pollution point sources as well as transportation, which would give a constant overview of the effect of air and water pollution impacts on living nature and of critical loads. The national monitoring programme should deliver data from pollution sources, end-of-pipe data, downstream to the coastal sea. In the long term, the quality of surveys could be raised by combining conventional analytical methods and surveys of bioaccumulation in polluted water-bodies and in the coastal sea.

The implemented Estonian national environmental monitoring programme of hazardous substances, which follows EU and HELCOM recommendations, covers all major problem areas, sites, and aspects on a national scale. Operational monitoring by companies, required by the environmental permit system, complements the national network and gives the opportunity for detailed assessment of trends in water-bodies. Nevertheless, the statistical power of the present sampling is weak; in particular, the temporal frequency should be increased. Databases and inventories of industrial chemicals and hazardous substances should be developed next years in addition to the extensive site surveys where sources are found. Further, as proposed by HELCOM, it could be fruitful to integrate source-oriented and load-oriented approaches, since both are lacking full-scale consistent data coverage (HELCOM 2004a). In the future, in the course of developments in environmental management and the implementation of integrated pollution prevention and control (IPPC), monitoring obligations will be shifted towards industrial companies. The public authorities remain in charge of national surveys, assessments, reporting, and inspection.

Acknowledgement: The project was financially supported by the Ministry of Environment of Estonia.

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Received 17 April 2004, accepted 24 November 2004

Chemosphere 60 (2005) 914–921

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Persistent organic pollutant patterns in grey seals (*Halichoerus grypus*)

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Received 25 August 2004; received in revised form 6 December 2004; accepted 14 January 2005
Available online 8 March 2005

Abstract

The aim of this paper is to examine the patterns of polychlorinated biphenyls (PCB) in the grey seals (*Halichoerus grypus*) from the Baltic, Northeast and Eastern England, and the St. Lawrence Estuary (Canada). In fact, the feeding habits of the ringed seal (*Phoca hispida*) include ingestion of major quantities of benthic crustaceans that might cause observed differences obtained in PCBs, whereas the grey seal feed mainly on fish. The profile (percent in mixture) of polychlorinated biphenyls (PCB) 101, 118, 138, 153, and 180, and the total of their concentrations in mg/kg lipid in grey seals from the Baltic, from Northeast and Eastern England, and from the St. Lawrence estuary (Canada), were examined by principal component analysis (PCA). When considering the possible effects of consuming seafood by the grey seal, it is necessary to characterize populations and individuals according to the amounts they consume, since populations in different parts of the world are likely to show big differences in their consumption of seafood. The patterns differ between juveniles and adult animals, but the gender of adults and geography do not appear to play a role.

Keywords: Polychlorinated biphenyls (PCB); Grey seal (*Halichoerus grypus*); Principal component analysis (PCA); The Baltic Sea

1. Introduction

The Baltic Sea is almost totally enclosed by land, and only connected to the North Sea by narrow and shallow straits around Denmark and Sweden. It takes decades for all the water in the enclosed area to be renewed (Roots, 1996). In addition, the Baltic Sea is the final destination of discharges and land run-off from many highly industrialized countries. Persistent organic pollutants (POPs) are a group of toxic and persistent chemicals the effect of which on human health and on the environment include dermal toxicity, immunotoxicity, reproductive effects and teratogenicity, endocrine disrupting effects and carcinogenicity. Currently, the movement from afar of PCB from southern sources outside Estonia is highly significant (Fig. 1).

Seals are the subject of present research because they live at the top of the food chain of the marine ecosystem, and accumulate many highly toxic compounds (Zitko, 1985).

The concentrations of toxicants in the seals of the Baltic Sea are studied (Blomkvist et al., 1992; Haraguchi

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doi:10.1016/j.chemosphere.2005.01.063

Fig. 1. PCB calculated depositions in the region of the Gulf of Livonia (data by Nordic, 1999; Agrell et al., 2001).

et al., 1992; Roots, 1995, 1996, 1999), but the results are not compared with the data and results of other regions. In fact, the feeding habits of the ringed seal include ingestion of major quantities of benthic crustaceans that might cause the observed differences in retained PCBs, whereas grey seal feed mainly on fish (Sjöderberg, 1975). In particular, further studies need to cover the comparison of juveniles and adult grey seals. Male grey seals require more specific research in future. It is necessary to continue studying the male grey seals, since females excrete part of their PCB load during the feeding period of the pups. Female grey seals with high PCB concentrations do not only harm their off-springs, but also according to the data provided by the data Pomeroy et al. (1996), they may not only feed their own pups, which makes studies more complicated.

The aim of this paper is to examine the patterns of polychlorinated biphenyls (PCB) in the grey seals (*Halichoerus grypus*) from the Baltic, Northeast and Eastern England, and the St. Lawrence Estuary (Canada).

2. Methods and materials

The Estonian coastal waters serve as the south eastern boundary of the regular distribution of the grey and ringed seal in the Baltic Sea. Of the above-mentioned two species, the grey seal is more abundant. The grey seals are mostly concentrated in the West-Estonian Archipelago Biosphere Reserve (WEABR) (Fig. 2). The WEABR is probably the best breeding area for seals in the Baltic Sea.

As seals are under wildlife protection in Estonia, the only way of collecting the samples is to use dead seals obtained from the nets of fishermen.

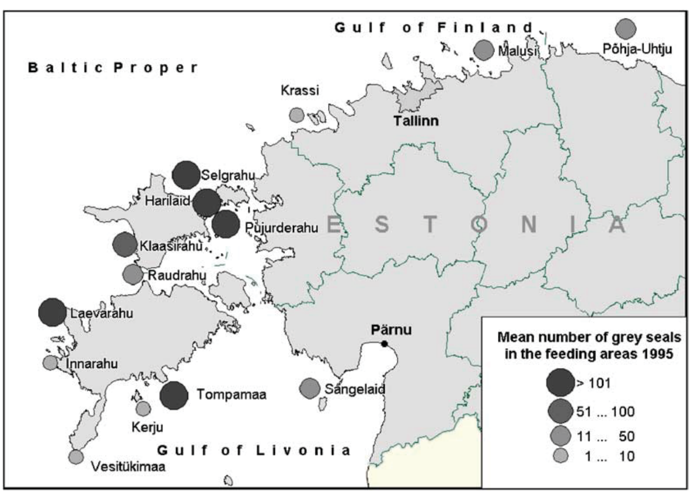

Fig. 2. Location of grey seal feeding areas in Estonian coastal waters (data by Estonian Environment, 1996).

Table 1
Profiles of chlorobiphenyls 101, 118, 138, 153, and 180 (in % of their sum) and their total absolute concentration, in grey seals (code: j—juvenile, M—male, F—female)

Author	No.	Code	Patterns[a]	101	118	138	153	180	Total conc. of PCB, mg/kg lipid
Haraguchi et al., 1992	1	j	J, B mean	2.08	0.86	40.74	40.74	15.58	41.73
	2	j	J, B mean	2.23	1.11	37.08	44.50	15.08	40.45
	3	M	AM, B	1.14	0.00	33.20	47.95	17.71	54.22
	4	F	AF, group A	1.60	0.00	33.16	44.92	20.32	93.50
	5	F	AF, group B	1.47	0.00	32.35	44.12	22.06	204.00
	6	F	AF, group C	2.33	0.00	34.19	45.08	18.41	266.20
	7	F	AF, group D	0.96	0.00	33.26	46.07	19.71	405.90
	8	F	AF, group E	0.73	0.00	32.92	47.39	18.96	1361.00
Bernt et al., 1999	9	M	AM	1.74	0.59	30.05	48.31	19.31	11.45
	10	F	AF	2.17	0.93	24.38	40.33	32.20	6.65
	11	j	JM[b]	4.97	1.65	31.03	45.88	16.46	2.86
	12	j	JF[b]	6.04	2.56	31.70	44.32	15.39	1.84
	13	j	M pup	4.92	1.75	29.28	43.37	20.67	3.25
	14	j	F pup	5.72	2.32	30.51	45.12	16.33	3.15
Law et al., 1989	15	j	Grey, Farne	3.92	5.63	35.78	42.59	12.09	7.07
	16	j	–	2.23	2.70	31.69	49.29	14.08	10.79
	17	F	–	2.85	3.98	25.71	42.28	25.17	2.04
Roots, 1996, 1999	18	F	2-3yF	0.87	0.68	31.32	40.83	26.30	8.83
	19	M	2-3yM	1.27	0.68	34.78	43.65	19.63	11.80
	20	M	6-8yM, Vilsandi	0.97	0.44	26.93	46.41	25.24	94.84
	21	M	6-8yM, Vilsandi	2.03	0.73	27.47	32.73	37.04	56.16
	22	M	5-6yM	1.49	1.14	33.40	42.01	21.97	6.85
	23	M	5-6yM	1.20	0.59	31.70	44.30	22.20	9.63
Vetter et al., 1995	24	F	19yF	3.08	2.39	26.92	40.34	27.27	1.17
	25	F	18yF	5.68	8.52	26.17	37.12	22.52	0.99
	26	F	15yF	6.64	11.73	28.76	34.29	18.58	0.45
	27	F	8yF	5.73	4.52	33.18	41.33	15.23	0.66
	28	j	JF	8.54	4.86	33.28	43.45	9.87	0.68
	29	M	26yM	1.73	3.23	26.65	44.66	23.74	3.47
	30	j	JM	7.26	10.20	31.75	37.42	13.38	0.44
	31	j	JM	7.14	5.95	34.52	39.88	12.50	0.17

[a] A—adult; M—male; F—female; J—juvenile; y—age of the seal.
[b] Nos. 11 and 12—pups and juvenile seals are of average of three years.

All solvents used were of the highest quality commercially available. Ten grams of the grey seal sample (muscle; blubber) were homogenized in a IKA T25 homogenizer (from Labassco AB, Pertille, Sweden) and extracted (*n*-hexane–acetone 1:1) according to Jensen et al. (1983) and the lipid content was determined by the method used by Jensen et al. (1983), Haraguchi et al. (1992) and by Roots (1996, 2001). Chromatography on a silica gel–concentrated sulfuric acid column removed lipid (0.1–0.2 g) from the extracts. The recovery of organochlorines from the extraction and clean-up procedures were measured.

For quantitative determination of PCB congeners, the internal standard IUPAC 189 was added. PCBs were analysed on a 100 m capillary columns (DB-5 or CPSil 8) using gas chromatography (Varian 3380) with electron capture detector (ECD). PCB isomers with IUPAC numbers 28, 52, 101, 105, 118, 138, 153 and 180 were analysed. The detection limit for different PCBs was around 1 μg/kg fresh weight (grey seal or fish muscle tissues). The Estonian Environmental Research Centre (EERC) is acknowledged by the German accreditation bureau Deutsches Akkreditierungssystem Prfwesen GmbH (DAP) (DAP-PL-3131.00 (2008-11-22)).

The concentrations of the five PCBs (IUPAC no. 101; 118; 138; 153; 180) in the seals blubber were scaled to a sum of 100, and the sum of their concentrations in mg/kg lipid weight was added as the sixth variable. The resulting data set (Table 1) was centred (mean of each variable = 0) and scaled (standard deviation of each variable = 1). The set was then examined by principal component analysis (see for example Zitko, 1994) by the programme of Wise and Gallagher (1998), running under Matlab 5.0 (The Math Works Inc., South Natick, MA 01760, USA).

3. Results and discussion

The Estonian coastal waters serve as the southeastern boundary of the regular distribution area of the grey (*H. grypus*) and ringed (*Phoca hispida*) seal in the Baltic Sea. During the annual moult period in May–June in the Estonian coastal waters, the stock size is established to be 1200–1500 grey seal individuals—that is, roughly 25% of the whole Baltic population (Fig. 2).

Comparing the PCB contents in the grey seal's blubber of Estonian coastal sea—WEABR (Väinameri Sea) (sample nos. 18, 19, 22 and 23) and Vilsandi (sample nos. 20 and 21), it appears that the PCB content in Vilsandi grey seal is higher but comparable or somewhat lower than in grey seals caught from the open Baltic Sea (Haraguchi et al., 1992). Very low fat percentage in blubber of 6–8 year-old male grey seal organism arose interest. Only individuals with a poor health status or nutrition status (thin layer of blubber and/or low content of extractable fat in blubber) had significantly higher concentrations of pollutants than other groups. By the data (Kalantzi et al., 2005) the PCB concentrations in the 1999 grey seal samples from the North Sea, showed a significant differences between the two different seasons—winter and summer. This trend is probably attributable to blubber loss rather than feeding habits (Kalantzi et al., 2005). Besides the above mentioned, the PCB concentrations in grey seals may also depend on migrations. There is quite little information about grey seal's present migrations in the Baltic Sea (Estonian Environment, 1996).

When considering the possible effects of consuming seafood by the grey seal, it is necessary to characterize populations and individuals according to the amounts they consume, since populations in different parts of the world are likely to show big differences in their consumption of seafood. In fact, the feeding habits of the ringed seal (*P. hispida*) include ingestion of major quantities of benthic crustaceans that might cause the observed differences in retained PCBs, whereas the grey seal feed mainly on fish. In the 1970s, when seals became a target of research, it was soon noticed that there was an alarming decrease in their reproductive capacity in the Baltic Sea populations. Comparing the food chain of the grey seal (*H. grypus*) at the beginning of 1970s (Sjöderberg, 1975) and species content during experimental catches in spring in the middle of the 90s (Kangur, 1996), we can assume that nowadays the grey seal do not eat so much cod, salmon, sea trout, etc. as they did at the end of 1970s. The research on the structure the grey seal's food carried out in the beginning of the

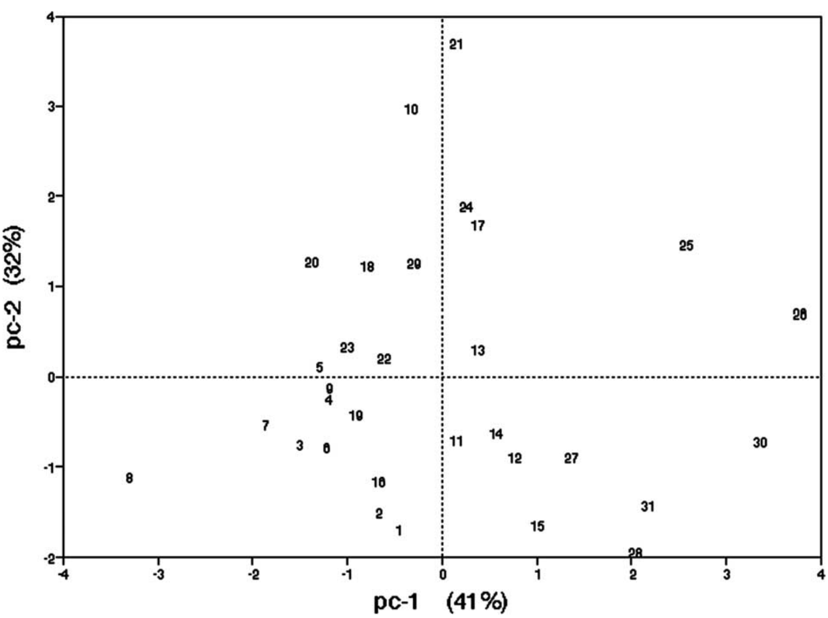

Fig. 3. Projections on the plane of the principal components pc-1 and pc-2. The fractions of the original variance captured by the pc's is indicated on the axes. For sample identification see Table 1.

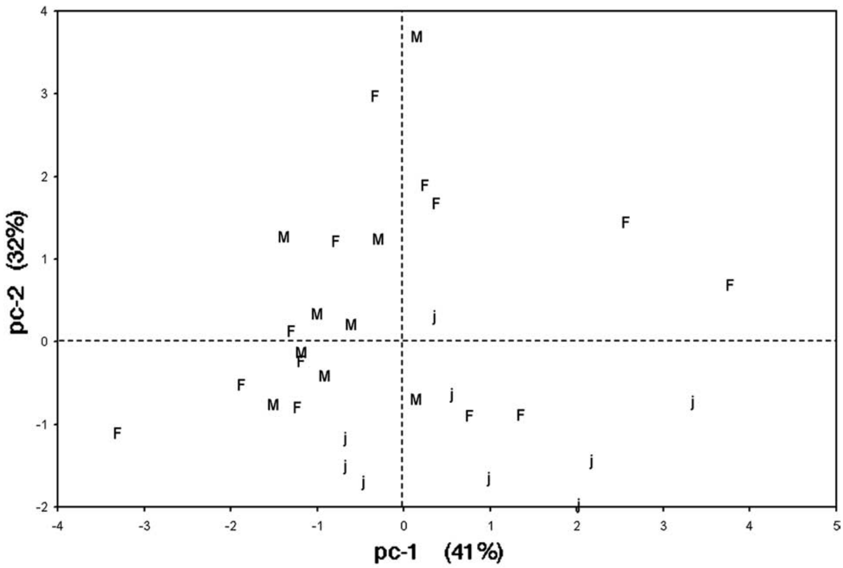

Fig. 4. Projections on the plane of the principal components pc-1 and pc-2. The fractions of the original variance captured by the pc's is indicated on the axes. Samples are identified by codes (see Table 1).

1970s mainly in the central part of the Baltic showed that the food contains 23.5% herring, 21% cod, 12.5% salmon, 7% sea trout, 5.6% eel and flounder, 4.9% perch and other fish in smaller amounts (Sjöderberg, 1975).

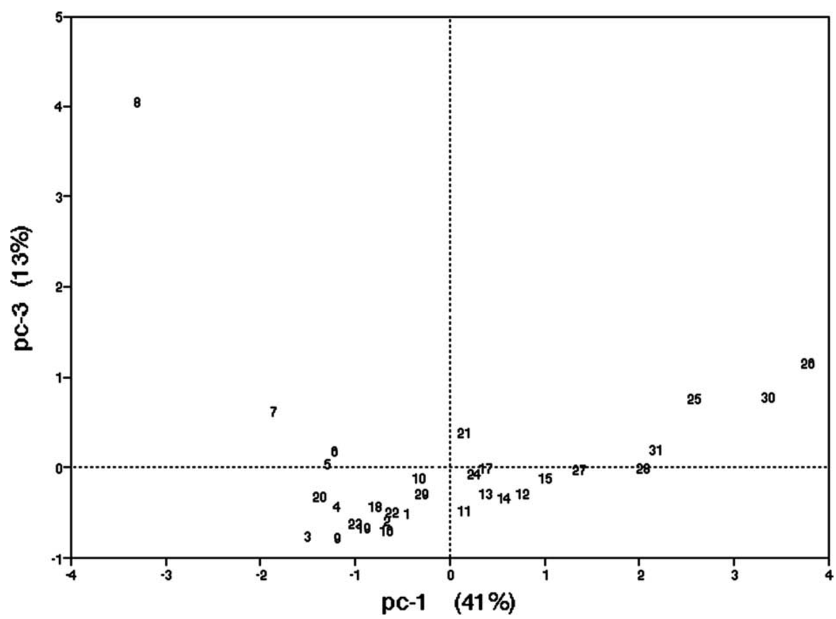

Fig. 5. Projections on the plane of the principal components pc-1 and pc-3. The fractions of the original variance captured by the pc's is indicated on the axes. For sample identification see Table 1.

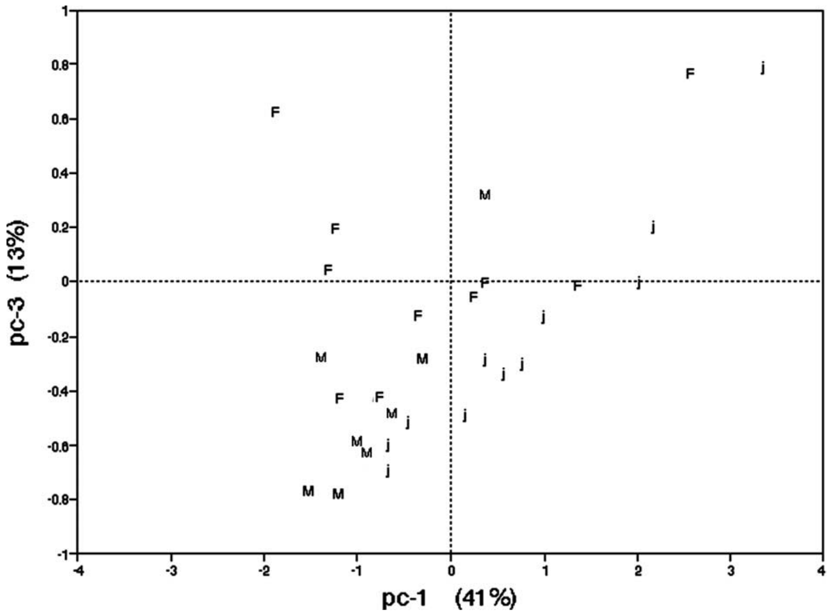

Fig. 6. Expanded part of Fig. 5. The sample 'F' in the upper left-hand corner is sample '7' in Fig. 5. Samples are identified by codes (Table 1).

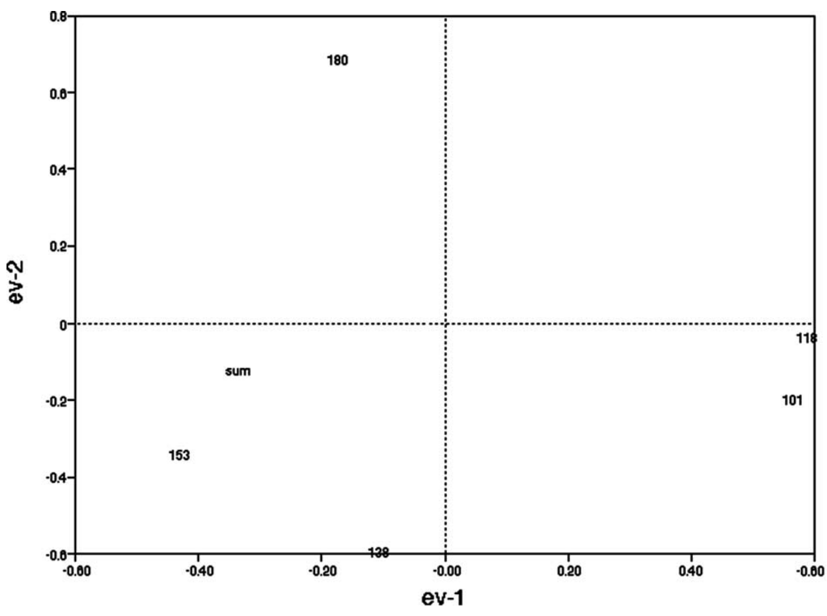

Fig. 7. Loading of the original variables (chlorobiphenyls 101, 118, 138, 153, and 180, and their total concentration ('sum')) on the principal components pc-1 (ev-1) and pc-2 (ev-2).

The decrease in the Baltic Sea water-salinity has had an essential effect on the ecosystem of the sea (especially on its northern and eastern parts) (Roots, 1996, 1999). The increase in the percentage rate of empty stomachs of

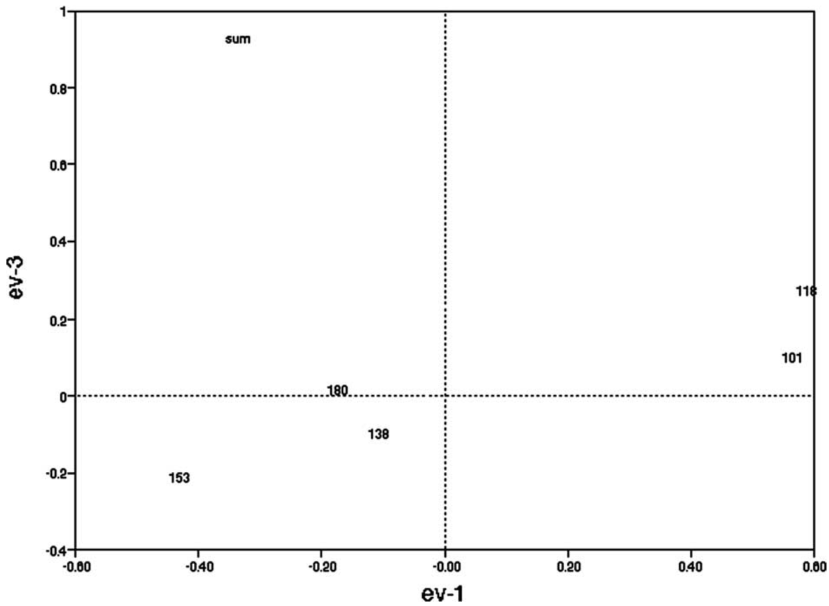

Fig. 8. Loading of the original variables (chlorobiphenyls 101, 118, 138, 153, and 180, and their total concentration ('sum')) on the principal components pc-1 (ev-1) and pc-3 (ev-3).

Baltic herring, sprat (Lankov and Kukk, 2002) and perch (Lappalainen et al., 2001; Roots et al., 2003) at the beginning of 1990s may turn out to be one of the reasons for the decrease of PCB concentrations in grey seal food, compared with the end of the 1970s (Roots, 1999).

In comparison with harbour or common seals (*Phoca vitulina*), there are relatively few data on chlorobiphenyls in grey seals (*H. grypus*) (Roots, 1996, 1999) and the aim of this paper is to examine the patterns of chlorobiphenyls in grey seals. The majority of papers, found in the literature, reports the chlorobiphenyls 101, 118, 138, 153, and 180. Most of the papers also list the concentration of the chlorobiphenyl 52, but this chlorobiphenyl was not included, to allow the use of the data by Vetter et al. (1995). Jenssen et al. (1996) presented the concentrations of only three chlorobiphenyls of the common set and their data on the concentration of chlorobiphenyls in neonatal pups could not be included. The first three principal components accounted for 86% of the original variance. As can be seen from Figs. 3–8, there are no 'clusters' in the data. However, the data contain several 'outliers', such as the samples 8, 10, 21, 25, 26, and 30 (Table 1). The reasons for the 'outlier' status can be determined by examining the effects of the original variables on the principal components (Figs. 3 and 4). One can see that the proportions of the chlorobiphenyls 101 and 118 affects pc-1 positively, whereas the concentration of the chlorobiphenyl 153 and the 'sum' have a negative effect on pc-1. Similarly, the chlorobiphenyls 138 and 180 affect pc-2 in opposite directions are independent of the other chlorobiphenyls (Fig. 3). The 'sum' is the main factor affecting pc-3 (Fig. 4). Thus, for example, the seal number 8 is an 'outlier' because of the very high concentration of all the chlorobiphenyls.

As can be seen from Fig. 4, juvenile seals are separate from adult animals. This is the result of the lower sum of the chlorobiphenyl concentrations and of the lower relative concentration of the chlorobiphenyl 180. The relative concentration of this chlorobiphenyl is lower even in the juveniles from the Baltic (numbers 1 and 2, Table 1), who contain a much higher sum of the concentrations in comparison to other juvenile seals. There does not appear to be a difference between male and female adult seals. Unfortunately, the set contains only a small number of males. Geographic areas also do not affect the chlorobiphenyl patterns. The sum of the chlorobiphenyl concentrations has decreased considerably since 1980s, but the relative concentrations have not changed considerably since. Interestingly, adult females of that year do not contain the chlorobiphenyl 118 (Haraguchi et al., 1992), but this may be an artefact of the analytical technique. By the authors Haraguchi et al. (1992) it is remarkable that chlorobiphenyl 118 was not even detectable in some samples. Adult ringed seal and adult female grey seal, all lacked the peak corresponding to chlorobiphenyl 118, that may indicate a specific mechanism of metabolism of this CB in these species.

4. Conclusions

The profile (percent in mixture) of polychlorinated biphenyls (PCB) 101, 118, 138, 153, and 180, and the sum of their concentrations in mg/kg lipid in grey seals (*H. grypus*) from the Baltic, Northeast and Eastern England, and the St. Lawrence Estuary (Canada), were examined by principal component analysis (PCA). The patterns differ between juveniles and adult animals, but the gender of adults and geography do not appear to play a role. The concentration of chlorobiphenyls in grey seals has decreased, but requires further monitoring. There is a dearth of data on the concentration of chlorobiphenyls in other organs and tissues, as well as on the concentrations of the methyl sulfones. As is usual for other species and chemicals, a relationship between the concentration of the chlorobiphenyls and the health status (Blomkvist et al., 1992) of the animals needs to further examination.

Acknowledgments

The Estonian Ministry of the Agriculture, the Research and Writing Initiative of the Program on Global Security and Sustainability of the John D. and Catherine T. MacArthur Foundation Grant and the Earl of Carlisle supported the project.

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Zitko, V., 1994. Principal component analysis in the evaluation of environmental data. Mar. Pollut. Bull. 28, 718–722.

CURRICULUM VITAE

Antti Roose

Date of birth, place: **8.7.1964, Kiviõli, Estonia**
Aadress: **University of Tartu, department of research and development, Ülikooli 18, 50090 Tartu**

Academic qualifications

2000–2004	University of Tartu, doctoral studies in geoinformatics
1993–1994	Manchester Metropolitan University, MPhil studies in GIS
1991–1993	University of Tartu, MSc in geography
1982–1989	University of Tartu, Diploma in geography

Employment history

2000–	project manager, University of Tartu, the department of research and development
1998–1999	project manager, Ordnance Survey / Estonian Land Board
1995–1998	broadcaster, head of department, Radio Free Europe/Radio Liberty
1992–1994	adviser, deputy governor, Ida-Viru county government
1990–1991	head of department, Ministry of Environment, Estonian Environment Information Centre
1985–1989	inspector, Water Management and Protection Board

Research qualifications

Geographic Information Systems

2002–2004	drafting monitoring module for the environmental state registry
1999–	optimisation of environmental monitoring network, applications of Mapinfo, Vertical Mapperi and Crimestat in air pollution, water management, landscape sciences, hazardous substances; environmental mapping
1998–1999	data flow modelling for the cadastre, drafting cadastral map model, compilation of legal framework for land reform, institutional development of land board
1993–1997	spatial analysis, environmental modelling

1993–1994	planning environmental infrastructures
1993–1997	MPhil: *Spatial Analysis of Impact of Industrial Change on Air Pollution*; GIS applications in Ida-Virumaal, thematic mapping, spatial analysis and analysis of environmental time-series, Mapinfo applications
1991–1993	MSc: *Data management and GIS applications for environmental-economic appraisal — a case study of the Estonian oil shale basin, spatial data model*, mapping, spatial analysis, IDRISI applications
1988–1989	diploma *Hydrological model of Kiviõli underground mine*

Environmental protection and monitoring

2003–2004	head of workgroup, amendments of law on environmental monitoring
2003–2004	expert, EU Phare/Twinning project *Development of Administrative Capacity for Monitoring and Evaluation of the Agri-environment Measures*
2002–2003	head of workgroup, EU LIFE Viru-Peipsi catchment area management project
2001–	environmental assessment of housing and ecological engineering in construction sector (Phare projects *Support System for Green Construction in the BSR* and *Institutional Capacity Building of BSR Industrial Research Institutes to support sustainable industrial development*), surveys on noise and working environment, methodologies on sustainable physical planning
2000–2004	member of Steering Committee, project *EU-approximation and institutional Strengthening of the Estonian Marine Monitoring System* (EISEMM), Estonia-Denmark
2000–	facilitation of environmental indicators, pilot studies on environmental statistics
1999–	surveys and studies on environment, tourism and renewable energy in Võrumaa
1999–2004	programming and strategic development of the Estonian national environmental monitoring programme
1995	expert, OECD survey *Financing Environmental Expenditure in CEE, Harvard Institute for International Development*
1992–1994	facilitator, the North-eastern Estonian Environmental Programme
1985–	environmental surveys in the north-eastern Estonia

Environmental project management

2001–	national co-ordinator, *Baltic 21 Institute for Sustainable Industry*
1999–2005	co-ordinator for the Estonian environmental monitoring programme
1998–1999	manager, EU Phare Land Reform project
1995	manager, COWIConsult project, *Financing for the environment in Estonia*
1992–1994	manager, EU Phare *Model Restoration Project on Kohtla Opencast Mine*

Scholarships

2003	International Association of Landscape Ecology, to attend IALE congress, Australia
1998	Training programme in the Ordnance Survey, Southampton, United Kingdom
1993–1994	The British Council, MPhil studies in the Manchester Metropolitan University
1993	Environmental programme of the United States Information Services, 2 months in USA
1992–1994	GIS project co-ordinator in Estonia, Baltic University Programme, Uppsala University, Sweden
1992	The Swedish Institute, 3 months master studies at the Stockholm University, Sweden

PUBLICATIONS

Peer-reviewed

Roose, A., Sepp, K., Saluveer, E., Oja, T.: Neighbourhood-defined approaches for integrating and designing landscape monitoring in Estonia. Landscape and Urban Planning — submitted 09.08.2004.

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Others

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Roose, A., 1994. Air pollution modelling in the Estonian Oil Shale Region. IGU Regional Conference. Prague, 22–26 August 1994. Proceedings on CD ROM.
Roose, A., 1994. Compounding effect of industrial change: a GIS application. In: 5th European Conference on GIS. Paris, March 29–April 1, 1994. Conference Proceedings. EGIS Foundation, pp. 988–996.
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Aadress: **Tartu Ülikooli teadus- ja arendusosakond,**
Ülikooli 18, 50090 Tartu

Haridus

2000–2004	Tartu Ülikool, doktoriõpe
1993–1994	*Manchester Metropolitan University*, MPhil
1991–1993	Tartu Ülikool, MSc
1982–1989	Tartu Riiklik Ülikool, Diplom

Töökohad

2000–	Tartu Ülikooli teadus- ja arendusosakonna projektijuht
1998–1999	*Ordnance Survey* / Maa-amet, projektijuht
1995–1998	Raadio Vaba Euroopa, toimetaja, osakonnajuhataja
1992–1994	Ida-Viru maavalitsus nõunik, asemaavanem
1990–1991	Keskkonnaministeeriumi Info- ja tehnokeskus, osakonnajuhataja
1985–1989	Kirde-Eesti Vee Kasutamise ja Kaitse Valitsuse inspektor

Kvalifikatsioon, teadus- ja arendustegevus

Geograafilised infosüsteemid

2002–2004	keskkonnaregistri seiremooduli koostamine
1999–	keskkonnaseire võrgustike optimeerimine, *Mapinfo*, *Vertical Mapper*'i ja *Crimestat* rakendused välisõhu seires, veeseires, maastikuseires ja ohtlike ainete seires; keskkonnakaartide koostamine
1998–1999	andmevoo mudeli arendamine maakatastris, katastrikaardi mudeli koostamine, maareformi rakendusaktide koostamine ja maakorralduse institutsionaalne arendamine
1993–1997	saastatuse ja keskkonnameetmete ruumiline modellerimine
1993–1994	infrastruktuuride planeerimine

1993–1997	MPhil: *Spatial Analysis of Impact of Industrial Change on Air Pollution;* GIS rakendused Ida-Virumaal, teemakaardistamine, ruumianalüüs ja keskkonnaandmete aegridade analüüs, Mapinfo rakendused
1991–1993	magistritöö "Andmeohje ja geoinfosüsteemide rakendusi keskkonnaökonoomilistes hinnangutes Ida-Virumaa näitel", ruumiandmemudeli ja digitaalkaartide koostamine, IDRISI rakendused
1988–1989	diplomitöö "Kiviõli kaevanduse hüdroloogiline mudel"

Keskkonnakaitse ja keskkonnaseire

2003–2004	keskkonnaseire seaduse töörühma juht
2003–2004	ekspert, EL Phare Twinning/Põllumajandusuuringute Keskuse projekt põllumajandusliku keskkonnatoetuse seire- ja hindamissüsteemi väljatöötamiseks
2002–2003	EL LIFE Viru-Peipsi vesikonna veemajanduskava projekti seire töörühma juht
2001–	ehituslahenduste ja hoonete keskkonnahinnangud, ehitiste energiaefektiivsuse uuringud, müra- ja töökeskkonnauuringud, säästva ruumiplaneerimise meetodite arendamine
2000-2004	Eesti-Taani mereseire projekti järelvalve nõukogu liige
2000–	keskkonnaindikaatorite väljatöötamine ja keskkonnastatistika pilootuuringud
1999–	keskkonna-, turismi- ja taastuvenergia uuringud Võrumaal
1999–2004	riikliku keskkonnaseire programmi koordineerimine
1995	projektiekspert, *OECD survey Financing Environmental Expenditure in CEE, Harvard Institute for International Development*
1992–1994	korraldaja, Kirde-Eesti keskkonnaprogramm
1985–	keskkonnauuringud Kirde-Eestis

Keskkonnaprojektide juhtimine

2001–	rahvuslik koordinaator, *Baltic 21 Institute for Sustainable Industry*
1999–2004	keskkonnaseire projektide koordinaator
1998–1999	Phare maareformi projekti juht Eestis
1995	COWIConsult projektijuht Eestis keskkonnakaitse rahastamise projektis
1992–1994	projektijuht, Phare Kohtla karjääri rekultiveerimismudeli projekt

Uurimistoetused ja stipendiumid

2003	Rahvusvahelise Maastikuökoloogia Assotsiatsiooni stipendium IALE kongressil osalemiseks, juuli 2003, Austraalia
1998	Koolitus *Ordnance Survey*'s, Southampton, Inglismaa
1993–1994	Briti Nõukogu, õpingud *Manchester Metropolitan University*'s
1993	*United States Information Services* keskkonnaprogramm, 2 kuud USA-s
1992–1994	GIS projekti koordinaator Eestis, *Baltic University Programme*, Uppsala Ülikool
1992	Rootsi Instituut, 3 kuud kraadiõpinguid Stockholmi Ülikoolis