The Integration of Scales in Landscape Ecology

Guest editors J.T.A. Verhoeven (Utrecht University)
M.J. Wassen (Utrecht University)
D.F. Whigham (Smithsonian Environmental Research Center/Utrecht University)
H. Middelkoop (Utrecht University)
G. van der Lee (WL Delft Hydraulics)


Foto: Saxifraga, Joep Dirkx

The Integration of Scales in Landscape Ecology

Scale issues are fundamentally important in all aspects of landscape ecology. The spatial scale of observation and analysis in landscape ecology concerns clusters of ecosystems and ranges from local to global. The last four decades have seen an enormous research effort at the ecosystem level, to understand biogeochemical cycles and food webs, explain the dynamics in biotic communities and populations, and find the mechanisms behind the general decline in biodiversity worldwide. More recently, there has been an increasing emphasis on studying ecological processes at the global scale. Understanding the dynamics of system earth over long time periods is necessary to be able to predict and finally influence the ecological effects of regional disturbances such as eutrophication of the coastal zone and more global disturbances such as climate change.

The landscape scale is of great importance for regional planning and for developing policies to influence global change. Understanding the processes responsible for our environmental quality not only requires a good knowledge of ecosystem functioning, but also a clear insight in the interactions among ecosystems in the landscape through water and matter flows, or migration of organisms or diaspores. In order to analyse ecological processes at a global scale, we need to be able to use reliable data sets compiled for whole catchments or regions. Integration of data collected at different scales is a prerequisite in these studies. In landscape ecology the importance of issues of scale has been recognized at an early stage. Problems and solutions regarding integration of scales have been explicitly addressed in many studies, and different approaches have been attempted.

This special issue of *Landschap* deals with this subject and contains six contributions which were presented during the seminar 'The Integration of Scales in Landscape Ecology' organized by the WLO and by Utrecht University on 13 June 2002. These contributions deal with the importance of identifying scale issues in landscape ecological studies and with methodologies to bring data collected at different scales to one common scale level for analysis by applying upscaling or downscaling approaches.

The contribution by Burrough and Pfeffer explains techniques for downscaling by using cheap, high-resolution data from digital elevation models to enhance the spatial resolution of mapped vegetation patterns in the Austrian alps. They reason that successful downscaling is only possible through the use of ancillary fine detail (e.g. high resolution remote sensing or digital elevation models), and process-based and empirical modelling (e.g. logistic regression or neural networks) based on substantial data sets.

Wassen and Verhoeven focus on up-scaling approaches. They emphasize that predictability depends on the relation between the spatial and the temporal scale of study. Three examples of scale dependent processes illustrate the importance of identifying the scale at which processes operate to avoid erroneous conclusions. They advocate that landscape studies should at least provide an explicit framework revealing differences in scale, since the questions asked have to be translated into spatial scenarios and subsequently into input maps.

Arheimer's contribution shows how several modelling techniques were combined in an approach of upscaling of data collected at the site scale to the scale of a region or a whole country. She uses the example of nitrogen leaching and transport from small subcatchments and describes how models of local nitrogen transport were linked to a set of nested models describing hydrological processes at different scales to finally estimate the total contribution of Sweden to the nitrogen loading of the Baltic Sea.

In a forum article, De Wit proposes an alternative approach for analysing and solving environmental problems at the river basin scale. Rather than the use of existing model studies and data sets which were developed for smaller scales and scaling these up to the river catchment scale, he advocates the explicit identification of the interaction framework at the scale of the policy questions, and the collection of new data and process information at this scale level if these are not already available. He illustrates his much simpler approach with examples for the rivers Rhine and Elbe.
The contribution of Whigham *et al.* deals with an attempt to evaluate the condition of wetlands in terms of functioning and biodiversity through the analysis of existing spatial geographical data in a GIS, rather than using the classical way to assess individual wetlands in a field-based approach. This new approach greatly enhances the opportunities to meet the increasing need for evaluating the condition of wetlands at the catchment scale. Whigham *et al.* used a statistical approach to compare both methodologies for a large region in the Chesapeake Bay area in the USA.
Finally, Mander *et al.* investigate the scale issues involved in territorial ecological networks. In their contribution, they demonstrate hierarchical aspects of such networks and analyse the opportunities and limitations for downscaling and upscaling of their functions. They discuss a number of principles which are helpful in understanding scale issues in ecological networks, i.e. connectivity, multifunctionality, continuity, and plenipotentiality.
As guest editors for this special issue, we hope that these contributions will give the readership of *Landschap* an overview of current discussions on the very important issue of scale in landscape ecology. We want to thank the authors for their excellent work and for their respect for the time schedule we had for this special issue.

JOS VERHOEVEN, MARTIN WASSEN, DENNIS WHIGHAM, HANS MIDDELKOOP, GUDA VAN DER LEE

How to find the landscape scale?

The concerted action of scientists to integrate scales in landscape ecology, may offer us an opportunity to determine the landscape scale more clearly and concisely define the term landscape. The definitions that we find in the literature all include wording that describe a landscape as a spatial unit that has the characteristics of an ecosystem. This raises the question why an ecosystem with spatially defined boundaries is not always a landscape. Why are continents, oceans or even the total biosphere not landscapes? This suggests that a landscape has a size below a certain maximum scale. Starting from the other end of the scale spectrum, we may conclude that a landscape must have a size that is above a critical minimum. A heathland pond or a calcareous outcrop have all of the characteristics of ecosystems with clear boundaries, but they would not be called landscapes.

A symposium on upscaling and downscaling could perhaps help us identify the upper and the lower scale limits for the term landscape. Systems ecology alone is not sufficient to offer the answer. That discipline goes up and down the total ladder of scales, without splitting off certain groups based on size. This means that landscapes cannot be spatially defined to only consist of the elements of an ecosystem: the physical and chemical components, the flora, the fauna and their interactions. The only component that is not yet included in the conventional definition of the ecosystem is man. This logic leads to the understanding that landscapes are spatially characterized by the action of man and the boundaries of landscapes are created by spatial differences in human actions. To find the specific scale of landscape on the total ladder of scales, should then take the home range of human populations as a starting point.

Etymology helps us at this point. Land is an ancient word for home; ‘scape’ comes from the Dutch word ‘schap’ which derives from ‘scheppen’ which means ‘to create’. Land-scape is: the home country that man created for himself. As a consequence, two different disciplines can be recognized: landscape ecology and ecology of landscapes.

In the first sense, landscape means the study of pattern and processes in ecosystems with no clear definition of space, thus offering a variety of scales. In the second sense, landscape means the study of the structure and the functioning of ecosystems that have man as a characteristic species. The boundaries of that system are found where populations of the human species created clear spatial structures that end up at the limit of their home range.

New methodological steps are needed in the study of landscapes in the context of the field of landscape ecology and they should be based on an integration of a number of fields of study including physical geography and biology, through cooperation with the human sciences of anthropology, historic geography, social geography and economy. These human home range sciences may help us to find the landscape scale that is effective in evaluating the effects of humans across the world and developing effective management strategies to meet the needs of nature and human societies. Upscaling and downscaling may help us to place the ecology of landscapes in the total scientific field of landscape ecology.

JACQUES DE SMIDT (President WLO)

Foto: Saxifraga, Jan van der Straaten

Opportunities and constraints of downscaling in environmental research

Downscaling
Alpine vegetation
Detrended correspondence analysis
Universal kriging
K-means

Spatial data concerning many aspects of landscape are collected at many levels of resolution, but if combined in numerical models or statistical classifications, they must be brought to a common spatial scale. This can be achieved by upscaling (fine to coarse) or downscaling (coarse to fine). This article explains how downscaling procedures using cheap, high-resolution data from digital elevation models enhance the spatial resolution of mapped vegetation patterns in the Austrian alps.

PETER BURROUGH & KARIN PFEFFER

Prof. dr. P. A. Burrough and **Dr. K. Pfeffer**, Utrecht Centre for Environment and Landscape Dynamics (UCEL), Faculty of Geographical Science, Utrecht University, Heidelberglaan 2, Postbox 80115, 3508 TC Utrecht.

When studying landscapes, and the biological, chemical, physical and anthropological processes operating in them, we frequently must deal simultaneously with the very small and the very large. For example, hydrogen ion concentration determines the base status (pH) of clay minerals, which are the result of rock weathering in past and present climates. The lithologic variation of clay minerals over the landscape depends on processes of erosion, transport and sedimentation that operate over many scales. In turn, these factors affect the storage and supply of nutrients to plant roots, thereby influencing the types, structures and patterns of vegetation which determine both the aesthetic and ecological qualities of the land at large (Figure 1). It is no wonder that landscape ecologists have much to discuss concerning the best way to approach their complex study object (Klijn, 2002). The signals that excite them depend very much on the tuning of their antennae to the patterns and processes they consider to be of most importance.

Because it is impossible to measure everything at all levels of resolution (in the limit, 100% sampling of soil or landform would destroy the object of interest!), landscape ecologists are forced to extend the information inherent in their samples and observations to other scales. Upscaling is the process of extending knowledge from small observation units (known in geostatistics as the support – see Burrough & McDonnell, 1998; Goovaerts, 1997) to units having larger areas; the reverse process of predicting local attributes from studies covering large areas is known as downscaling (e.g. Bierkens et al., 2000; Canon & Whitfield, 2002; Sailor & Li, 1999).

Many aspects of landscape ecology involve upscaling from data about objects smaller than people to objects that are very much larger than people. Upscaling frequently requires interpolation or the use of numerical models to extend the knowledge obtained at point or local observations to the landscape at large. In other situations, which are becoming more frequent thanks to large amounts of data in digital geographical information systems (GIS), we may have more information about the landscape over large areas and need means to extend or combine these data to make statements about local conditions. As already indicated, this is known as downscaling. The aim of this paper is to explain and illustrate how statistical methods of downscaling can enhance the value of expensive-to-measure data having a coarse (and possibly incomplete) spatial coverage through combination with cheap, readily available data having a finer spatial resolution.

Reasons for downscaling

Downscaling is the process of reconstructing fine detail from a general picture. This is a common issue in many Global Change studies, when General Circulation Models (GCMs) are used to predict climate-induced responses of local or regional hydrological conditions (Sailor & Li, 1999). Alternative means are necessary to predict local climatic changes at higher levels of spatial and temporal resolution (e.g. Cannon & Whitfield, 2002).

Although most pioneering research on downscaling comes from the Global Change community, the same

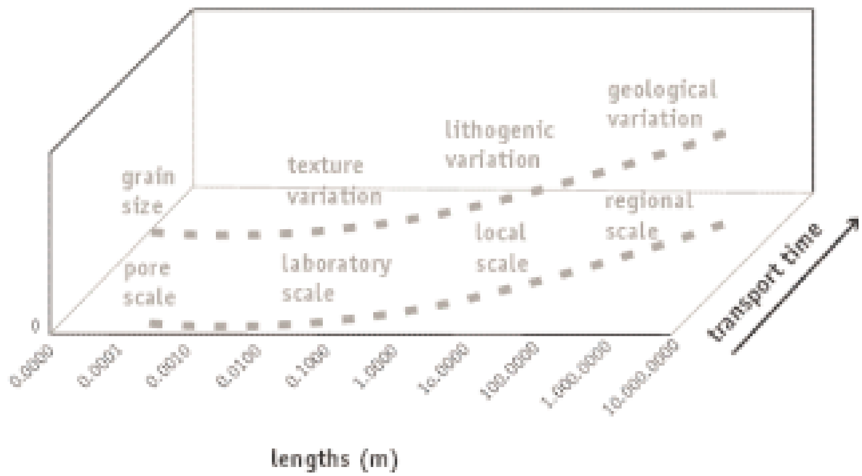


Figure 1. A schematic overview of the range of spatial scales encountered in studies of the physical landscape (adapted from Burrough 1996)

principles apply in landscape ecology when one attempts to predict aspects of the short-range spatial variation of vegetation within larger areas for which only generalised maps or sample surveys are available. For example, in landscape ecological studies it is not uncommon to want to predict the ecological condition of a small vegetation plot from generalised information over a whole region. This may be necessary for many reasons. Commonly occurring situations are:

- the sources of data have fixed levels of resolution that are too coarse for the application (e.g. attempting to infer details of individual patches of vegetation from remotely sensed imagery having 1 x 1 km pixels),
- numerical models of environmental processes often require data to be brought to a common level of spatial resolution,
- it is difficult to sample an area uniformly because of varying ease of access,
- data are sparse or incomplete.

There is much interest in downscaling the coarse resolution digital data obtained by remote sensing or climate models so that they may be linked to regional or local data when required. In recent years there has also been progress in bringing together international digital data sets that can be stored, displayed, analysed and combined in Geographical Information Systems – GIS – (Burrough & McDonnell, 1998; Burrough & Masser, 1998; Longley *et al.*, 2001). Drawing on developments in the United States, Europe and international organisations, Global Spatial Data Initiatives (GSDI) have lead to the establishment of digital data sets of elevation, climate, vegetation, hydrological basins, etc. that have commensurate levels of spatial (but not temporal) resolution (Figure 2). Many of these data sources are linked to standard cartographic map scales that imply a smooth transition in resolution from one level to another.

One of the most important recent developments in GIS technology has been the improved availability of high resolution digital elevation models (DEM). Today, it is quite possible to obtain DEMs of large areas of land with a spatial resolution that is finer than 5 x 5m. To give the reader

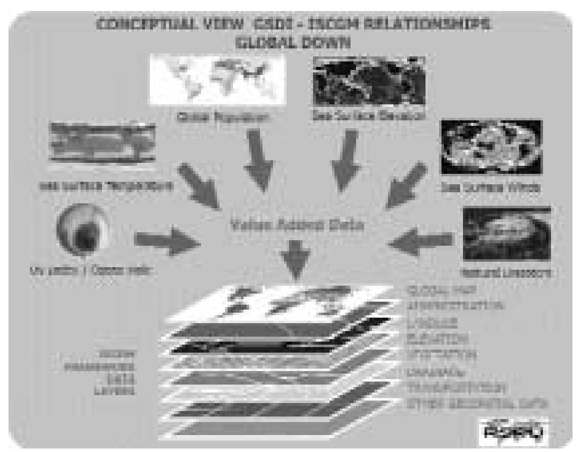


Figure 2. Shared global data may improve understanding of spatial processes affecting the planet, but only at the world scale. This figure and more details from:*http://www.iscgm.org/html4/index.html*

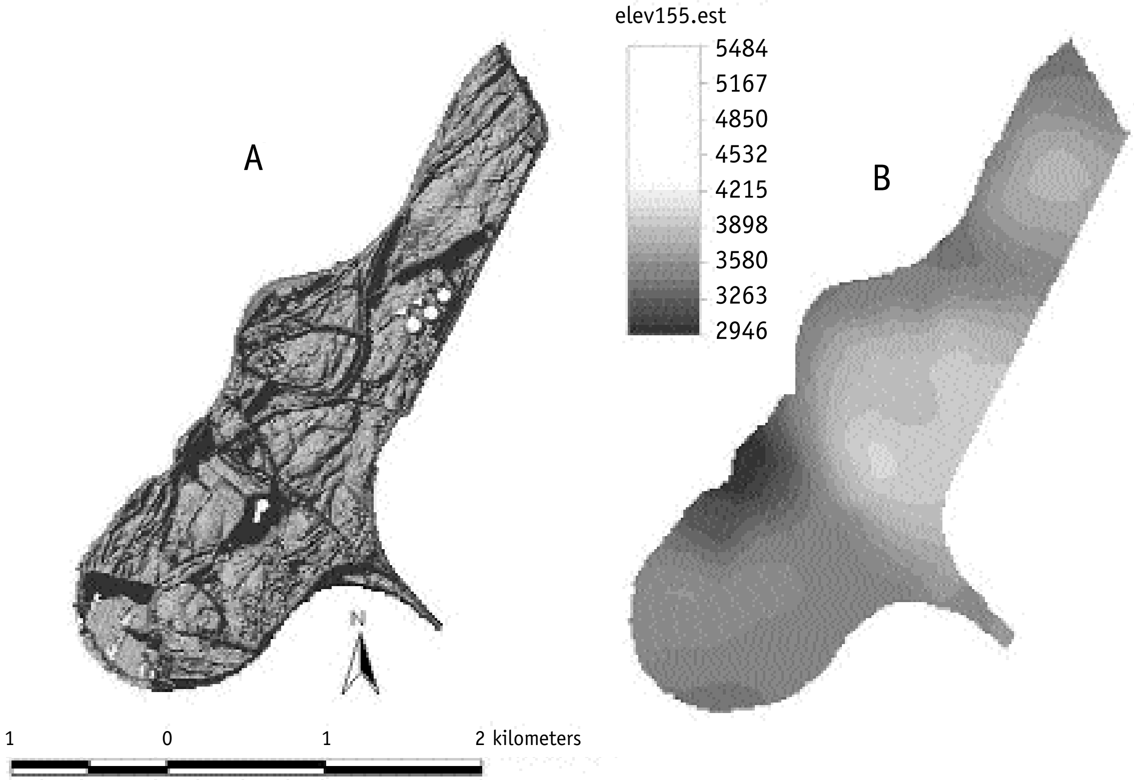


Figure 3 A comparison of elevation data (mm above local reference) obtained from Laser altimetry of part of the Maas floodplain , (courtesy Dutch Meetkundige Dienst) and interpolation by kriging. Left: 5 x 5m resolution, right: surface interpolated from 155 surface measurements to a grid of 20 x 20m. Clearly, the high resolution surface (left) gives much more information over surface structures and ecological differences than the low resolution surface (right).

an idea of the level of surface detail that is possible today, Figure 3 illustrates this for a part of the floodplain of the river Maas in a southern province(Zuid Limburg) in the Netherlands. From this figure we see that not only can elevation differences be computed directly over short distances, but also many ecologically relevant derivatives such as local slopes, aspect and direct received solar radiation and local drainage situations (Burrough & McDonnell, 1998).

As we know that many ecological processes in the landscape are moderated by differences in elevation, slope or incident solar radiation (Burrough *et al.*, 2001) a GIS can be used to calculate the derivatives of a DEM at any required level of spatial resolution, thereby providing a rich source of information on the possible short and long range spatial variation of ecological conditions. If the generalised, or expensive-to-measure attributes of vegetation types or landscape or regional climate can be linked to these detailed data, we have a means to downscale them to the fine level of detail provided by the DEM. Essentially, the global data will be modified by local variations in correlated secondary attributes to provide the more detailed downscaled picture. This can be achieved by using statistical methods and interpolation (Bierkens *et al.*, 2000; Sailor & Li, 1999).

The principles of downscaling

Figure 4, (modified from Bierkens *et al.*, 2000), illustrates the geostatistical principles of downscaling. The term support is used to indicate the size of the basic spatial unit for which data for a given attribute z are available.

The horizontal axis gives the size of the support s_i while the vertical axis gives the value of the regionalised variable z_i for the whole of that support. The size of support s_2 is the smallest spatial unit for the generalised data; within this basic unit the value of z is taken to be uniform be-

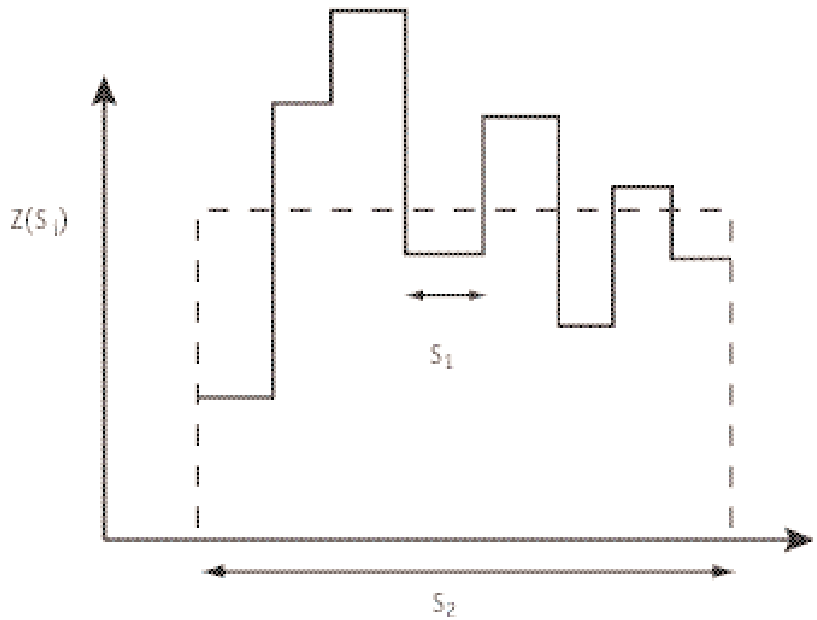


Figure 4. The principles of downscaling. Given data with the spatial resolution of s_2, reconstruct the variation of the attribute z for spatial resolution s_1

cause there is no more information. In other words, when the support is large (s_2) there is no information about the spatial variation of z within the dimensions of s_2 – only a mean value is known.

The size of the smaller support s_1 represents the desired level of spatial resolution. By downscaling we are attempting to create information about the more detailed variation of z. In this case the resolution of s_1 is eight times better than support s_2.

It is easy to generalise data from a fine to a coarse support. For a given cell there are many ways to compute the upscaled value of z_2 from the 8 data of z_1, the most obvious being the mean, or the mode, the median, the most commonly occurring value and so on. Downscaling – i.e. computing the values of the z_1 data from the z_2 is much more difficult. Because the same value of this s_2 mean can be obtained from a very large variety of, and operations on, the 8 values of the s_1 data: the variation of $z(s_1)$ shown is but one possible combination from an infinite set of possibilities based on the support s_1. This phenomenon, called equifinality, means that determining unique s_1 values from the s_2 value is impossible without extra information, so, given that we have information on z at the level of s_2, how can one predict z at the level of s_1?

There are two main approaches to downscaling that use various forms of regression:

- Have local, but sufficient amounts of empirical data on z at the level of s_1,
- Use large amounts of cheap, proxy data to predict z at the level of s_1.

Local, but sufficient amounts of empirical data on z at the level of s_1

Given sufficient amounts of data on z at the level of s_1, in principle we can use methods of spatial autocorrelation and interpolation (geostatistics) to estimate the spatial covariance of z for any required level of resolution (Burrough & McDonnell, 1998; Goovaerts, 1997; Heuvelink & Pebesma, 1999). Alternatively, through methods of conditional simulation, we may create models of the statistical nature of the spatial variation of z at the level of s_1. These models of spatial autocorrelation may be extended to areas for which we have none or very few data at the level of s_1 (e.g. Lagacherie *et al.*, 1995).

Use proxy data to predict z at the level of s_1

Proxies are attributes that are easier to measure than those about which information is desired, but which are thought to have a strong correlation with them. A well known example is the oxygen isotope ratio in ice cores, which is thought to provide a strong indication of climate change. As noted before, detailed digital elevation models may provide useful proxies for ecological variations in a landscape. Their value may be enhanced if they can be

combined with information on the probabilities of particular relations that are known to occur.

There are many other computational tools to convert spatial data from one level to another. Besides the methods of spatial autocorrelation already mentioned, these include process models (e.g. hydrological models, crop yield models, etc.), and empirical models based on logistic regression (e.g. Barendregt *et al.*, 1993), multivariate classification (Burrough *et al.*, 2001; Pfeffer, 2003), neural networks (Cannon & Whitfield, 2002) and similar approaches. Van Horssen *et al.* (1999) combined geographical information systems, geostatistical interpolation (kriging) and logistic regression modelling to predict plant species in wetland ecosystems in the Netherlands. Bierkens *et al.* (2000) and Burrough & McDonnell (1998) provide more details of these and other methods.

In a flat landscape, the values of the attributes of interest or their proxies are usually directly linked to the support in question. In mountainous and hilly landscapes, the data collected for any given instance of the support s_j may also depend on other factors. Note that with certain kinds of proxy data (e.g. derivatives from digital elevation models and reflected electromagnetic radiation detected by remote sensors), the attributes of an instance of a given support may vary depending on the geometrical orientation of the sampling grid (Demargne, 2001). Nevertheless, we ignore this complicating issue here.

A case study: downscaling Alpine vegetation data by a factor of 10 using a digital elevation model, detrended correspondence analysis, universal kriging and k-means clustering

Although it will be clear from the foregoing that there are many ways to achieve a downscaling of environmental data from the generalised to the particular level, we will attempt to elucidate the process further using a case study taken from recent practice (see Pfeffer, 2003, Pfeffer *et al.*, 2003). The example chosen concerns the need to carry out rapid mapping of vegetation in difficult to reach, high altitude areas of the Austrian alps that are much used for skiing so that the impact of the sport has a minimal effect on the natural alpine vegetation. Local planning for optimising the location of ski runs in mountain areas requires detailed spatial information on site factors such as vegetation, which is commonly lacking in rugged terrain. The direct sampling of vegetation in high altitude alpine areas is only possible for a limited period of the year and access is difficult so systematic mapping is expensive and rarely carried out. In high altitude alpine areas the collection of data from 10 x 10m quadrats on a 100m grid would be regarded as ‘detailed’, though it is clear from recent research that important vegetation differences may occur over much shorter distances in the alpine environment (Guisan *et al.*, 1998; 1999; Guisan & Zimmermann, 2000; Hoersch *et al.*, 2002).

In contrast to the difficulties of visiting many sample sites, the diversity of alpine flora almost guarantees the recording of large numbers of different plants, leading to a richness of information about plant communities, but little about their spatial patterns. Therefore we may have relatively much information about the composition of different plant communities, and relatively little about their spatial distribution. In these circumstances it makes

sense to use the quadrat samples to develop an optimal (i.e. the best local) classification for the vegetation data and to use a cheap proxy for spatial mapping (c.f. van Horssen *et al.*, 1999).
As noted above, current GIS technology makes it easy to create detailed digital elevation models from large scale (1:25000) digitised contour maps or aerial photographs. These topographic attributes and their spatial derivatives (slope, slope curvature, direct received radiation and wetness indices) are realistic ecological proxies for the supply of energy, moisture and nutrients that may influence plant growth and vegetation types (Burrough *et al.*, 2001; Hoersch *et al.*, 2002): they can easily be computed from a gridded digital elevation model (DEM) at any desired resolution. As explained in the following sections, the high resolution, cheap data were combined with the vegetation classes to map the short-range spatial variations of vegetation in the terrain.

Study area

The study area is located in the Ötztal, a north-south valley in the Tyrol, on the upper western slopes of the village of Sölden, which is a popular ski area in the Austrian Alps. It covers an area of approximately 3.6 km2, and has an elevation range from the timberline, at about 1900m, up to 2650m. Figure 5a shows a general view of the upper part of the study area, while Figure 5b shows short-range vegetation across narrow (20-50m) valley heads in the lower, east-facing part. Full details of the study area are given in Pfeffer (2003).
The procedure was as follows:

Vegetation sampling

During the summer of the year 2000, plant species occurrence was recorded at 223 quadrats, each 10m x 10 m, located on a reference grid of 100m x 100m (Figure 6a). In each quadrat all species were recorded according to ordinal abundance: 1 indicates the presence of a plant species, 2 means frequent occurrence and 3 means that a certain plant species was dominant. In total 147 species were identified, neglecting some grass species and all fungi and ferns. Fifteen quadrats were rejected because they fell on tracks or other disturbed ground leaving 208 for analysis.
The vegetation data show that the study area contains many common species, known to be typical for alpine grassland and alpine heaths (Reisigl & Keller, 1987). Although each species has its own preferences, some are broadly tolerant making it difficult to identify an unambiguous correlation of species preferences and ecological attributes. Certain key species were recorded which were characteristic for sites with specific conditions like a certain elevation range, exposure or moisture content. Although these key species are important for mapping vegetation types, they frequently occurred in narrow valleys with different conditions that were too small to be resolved by the 100 x 100m sampling grid. Therefore we sought a way to downscale these vegetation data so that the vegetation types occurring in the smaller components of the landscape could be predicted.
The first step in downscaling was to reduce the 208 x 147 vegetation site/species data matrix to manageable proportions. We used detrended correspondence analysis (DCA - Canoco 4.02: Ter Braak & Smilauer, 1998), which returned four axes with a cumulative explained variance that was only 20% of the total of the complete data set (Pfeffer *et al.*, 2003). This result suggests that much of the area is indeed poorly differentiated (i.e. it is covered by a broad range of similar species with a wide range of tolerance) and that rare species, if any, occur in the less frequently sampled parts of the landscape.

Figure 5. a: (Top) view of Hoch Sölden to the north; b: (bottom) west-facing low lying gullies with large variation of vegetation over distances of 20-50m

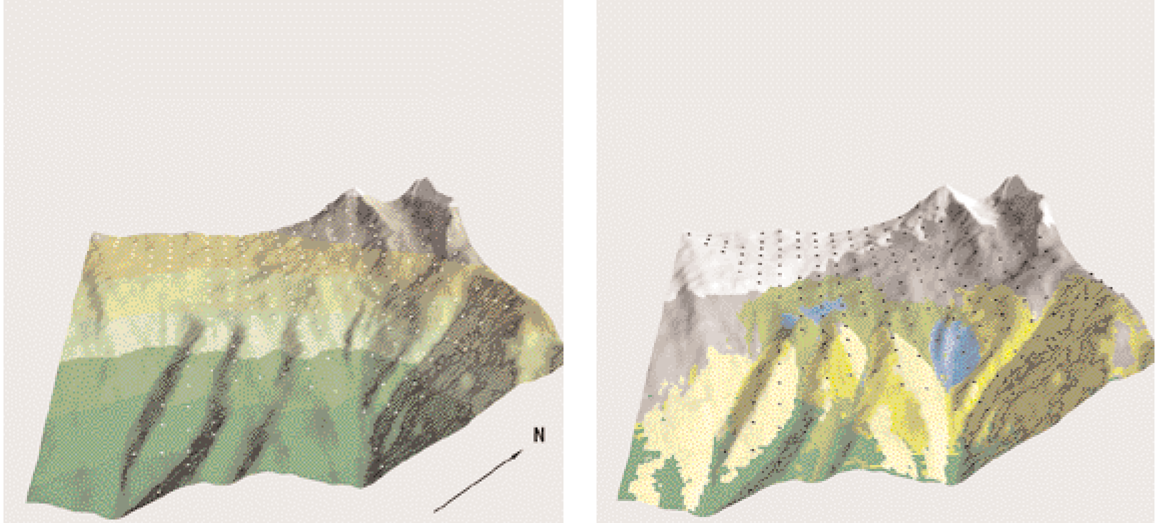


Figure 6. View from the west: a) Sampling network for 100 x 100m survey of vegetation (left): b) final vegetation classes mapped to 10m resolution by downscaling (right).

Creating high resolution proxies for mapping vegetation

We used a digital elevation model with cell sizes of 10m x 10m, (source: Bundesamt für Eich- und Vermessungswesen, Austria), which was the level of spatial resolution required for the downscaled vegetation map. The ecological proxies for vegetation namely altitude, slope, planform curvature, profile curvature, potential received annual solar radiation, distance to ridges, and mean wetness index were derived from the digital elevation model using PCRaster (PCRaster, 2002; Van Dam, 2001; Wesseling *et al.*, 1996). All results were stored in raster maps having a grid cell size of 10 m.

The downscaling procedure has four steps:

1 Compute the regressions between the dependent vegetation scores (DCA axes) and the independent proxies (elevation, slope, incident radiation, etc.) for the 208 quadrats.
2 Examine the residuals from these trends for spatial correlation using semivariogram analysis.
3 For each DCA axis, use the regressions and the semivariograms to create four DCA score maps at the resolution of the DEM.
4 Create 7 vegetation classes using a k-means classification of the original 208 DCA scores; use the k-means to allocate all points on the 10 x 10m grid to a vegetation class at the fine level of resolution desired.

Step 1 yielded the results given in Table 1, which confirm the assumed links between topographic proxies and vegetation scores, and provide the regression models (see Pfeffer, 2003).
Step 2 resulted in four spherical semivariogram models being fitted to the residuals from regression (Table 2). Parameter c0 indicates the level of non-spatial noise, c1 gives the level of spatially correlated variation, and a gives the range in metres over which that variation acts. The relations of c1 to c0 show the strong spatial dependence in all four sets of residuals, particularly for the first and third DCA axes.
Step 3 involved using the regression models and the semivariograms of residuals to interpolate each DCA score by universal kriging (Burrough & McDonnell, 1998; Goovaerts, 1997) to all cells on the 10 x 10m grid for the whole of the study area. This yielded 4 maps, one for each DCA axis.
In *step 4* k-means clustering first created 7 vegetation classes based on the DCA scores from the 208 sampled quadrats. The k-means clustering algorithm (Hastie *et al.*, 2001; MacQueen, 1967) is an iterative descent clustering technique designed to distribute multivariate data among k clusters, where k is typically less than 10 groups. For quantitative variables using a Euclidean distance metric, the total cluster variance is minimized with respect to the cluster means by assigning each observation to the closest mean. The means are recalculated and the observations are reallocated to the nearest clusters; this procedure is it-

erated until cluster memberships are stable.
Once the clustering had been carried out, all 10m x 10m grid cells were allocated to a class based on their interpolated DCA scores. The final map was displayed draped over the DEM for clarity (Figure 6b).

Dependent variable	Independent variables (proxies)	Multiple R^2
DCA1	Elevation, slope, incident solar radiation	0.7466
DCA2	Mean wetness index, elevation, slope	0.1095
DCA3	Incident solar radiation	0.2733
DCA4	Profile curvature	0.0221

Table 1. Main dependent variables contributing to each vegetation axis

Discussion and conclusions

The exercise reported in this paper demonstrates that even with noisy data and many plant species tolerant of a wide range of conditions, it was possible to downscale information from a relatively coarse vegetation survey to a much finer spatial resolution. This was thanks to the extra information obtained from geostatistical interpolation aided by simple proxies derived from a high resolution DEM. Field checking, particularly in the narrow valleys to the east of the study area, showed that in these limited areas the mapped vegetation, which was based on a very sparse sample of less than 10 quadrats, corresponded with the impression of the vegetation obtained in the field. The consistency analysis indicated that it was essential to include all kinds of vegetation type in the initial sample, especially if the vegetation type represented was not common.
We conclude that although downscaling has many limitations, the availability of cheap, spatially well-correlated proxies supported by regression and spatial autocovariance studies (i.e. universal kriging) may make it possible to create useful and detailed maps of vegetation types from sparse, expensive data.

Downscaling: opportunities and constraints

As the case study shows, downscaling is not simple and requires considerable understanding of the methods of data processing being undertaken. There are both opportunities and constraints, however. The opportunities include:

- the extension of knowledge from general levels to local detailed areas,
- methods can be automated,
- enables quick and reproducible coverage of large areas if properties are similar,
- downscaling makes good use of the available ancillary data and proxies, whether in mechanistic models or empirical functions.

The constraints include:

- an almost total lack of unique solutions,
- information that has been lost cannot be created from nothing – if a particular vegetation type has not been sampled then there is no information to link to fine scale proxies,
- many predictions will be based on stochastic relations that may be poorly understood,
- any single means of downscaling may not apply over all levels of the phenomena hierarchy (atoms to oceans),
- non-linearity and feedback loops may obfuscate the relations between emergent properties and details, or complexity and simple interactions,

Dependent variable/Parameter	c0	c1	a
DCA1	0.11	1.33	10823
DCA2	0.42	0.80	612
DCA3	0.48	3.49	12779
DCA4	0.58	1.23	2522

Table 2. Parameters of spherical model semivariograms fitted to the residuals of each DCA axis

- the quality of the regression models used in downscaling may be quite sensitive to relatively small variations in the size and composition of the data set. For example, omitting only a few sample sites from critical narrow valley sites resulted in a much poorer performance when downscaling the vegetation patterns of the case study area.

Abstract

Sizes of discernible spatial units in landscapes (called the support in geostatistics) range from very small ($<10^{-6}$ m^2) for soil particles and bacteria to very large ($>10^9$ m^2) for geological formations and climatic zones. Many environmental models require data at common levels of spatial resolution but it is clearly impossible to measure everything at either one, or all scales. Therefore, people attempt to link data collected at different scales either by predicting the attributes of large areas from sets of local, high resolution data (upscaling), or by inferring the attributes of small areas from generalised data on large areas (downscaling). Downscaling attempts to reconstruct the fine picture from regional patterns, but this may be achieved in an infinite number of ways.

Successful downscaling is only possible through the use of ancillary fine detail (e.g. high resolution remote sensing or digital elevation models), and process-based and empirical modelling (e.g. logistic regression or neural networks) based on substantial data sets of useful proxies or mechanistic, physically-based models. In this paper, downscaling is illustrated by an example from the Austrian alps in which detailed digital elevation models, universal kriging and multivariate clustering were used to improve the spatial resolution of high altitude, sparsely sampled vegetation patterns.

References

Barendregt, A., M.J. Wassen & J.T. De Smidt, 1993. Hydroecological modelling in a polder landscape: a tool for wetland management. In: Landscape Ecology of a stressed environment, C.C. Vos & P. Opdam (eds), Chapman & Hall, London, pp 79-99.

Bierkens, M.F.P., Finke, P.A. & P. de Willigen, 2000. Upscaling & Downscaling methods for Environmental Research. Developments in Plant & Soil Sciences Volume 88, Kluwer Academic Publishers, Dordrecht, 190pp.

Burrough, P A., 1996. Opportunities and limitations of GIS-based modeling of solute transport at the regional scale. Chapter 2 in D.L. Corwin & K. Loague (eds), Special SSSA Publication No. 48. Application of GIS to the Modeling of Non-Point Source Pollutants in the Vadose Zone. pp 19-38.

Burrough, P.A. & R.A. McDonnell, 1998. Principles of Geographical Information Systems. Oxford University Press, Oxford.

Burrough, P.A. & I. Masser, 1998. European Geographic Information Structures. Taylor & Francis, London.

Burrough, P.A., P.F.M. Van Gaans & R.A. MacMillan, 2000. High-resolution landform classification using fuzzy k-means. Fuzzy Sets & Systems, 113, 37-52.

Burrough, P.A., J.P. Wilson, P.F.M. Van Gaans, & A.J. Hansen, 2001. Fuzzy k-means classification of topo-climatic data as an aid to forest mapping in the Greater Yellowstone Area, USA. Landscape Ecology 16, 523-546.

Cannon, A.J. & P.H. Whitfield, 2002. Downscaling recent streamflow conditions in British Columbia, Canada using ensemble neural networks. Journal of Hydrology 259, 136-151.

Demargne, J., 2001. Qualité des Modèls Numérique de Terrain pour l'Hydrologie. Doctor's thesis, Université de Marne-la-Vallee, 29 June 2001.

Goovaerts, P., 1997. Geostatistics for Natural Resources Evaluation. Oxford University Press, New York.

Guisan, A., J.P. Theurillat, & F. Kienast, 1998. Predicting the potential distribution of plant species in an alpine environment. Journal of Vegetation Science 9, 65-74.

Guisan, A., S.B. Weiss, & A.D. Weiss, 1999. GLM versus CCA spatial modelling of plant species distribution. Plant Ecology 143, 107-122.

Guisan, A. & N.E. Zimmermann, 2000. Predictive habitat distribution models in ecology. Ecological Modelling 153, 147-186.

Heuvelink, G.B.M.. & E.J Pebesma, 1999. Spatial Aggregation and soil process modelling. Geoderma 89, 47-65.

Hastie, T., R. Tibshirani & J. Friedman, 2001. The Elements of Statistical Learning, Springer Series in Statistics, Springer New York.

Hoersch, B., G. Braun, & U. Schmidt, 2002. Relation between landform and vegetation in alpine regions of Wallis, Switzerland. A multiscale remote sensing and GIS approach. Computers, Environment & Urban Systems 26, 113-139.

Horssen, P.W. van, P.P. Scot & A. Barendregt, 1999. A GIS-based plant prediction model for wetland ecosystems. Landscape Ecology 14, 253-265.

Klijn, J. 2002. Kort & klijn. Landschap, 2002/1, 57.

Lagacherie, J.P. Legros & P.A. Burrough, 1995. A soil survey procedure using the knowledge of soil pattern established on a previously mapped area. Geoderma 65, 283-301.

Longley, P.A., M.F. Goodchild, D.J. Maguire, & D.W. Rhind, 2001. Geographic Information Systems and Science. J. Wiley, Chichester.

MacQueen, J.B., 1967. Some methods for classification and analysis of multivariate observations, In: Proceedings of 5th Berkeley Symposium on Mathematical Statistics and Probability 1, 281-297. University of California. PressBerkely, CA.

PCRaster, 2002. PCRaster Software. See http://www.geog.uu.nl/pcraster.nl.

Pfeffer, K., 2003. Integrating spatio-temporal environmental models for planning ski runs. PhD Thesis, Faculty of Geographical Sciences, Utrecht University.

Pfeffer, K., E.J. Pebesma & P.A. Burrough, 2003. Mapping alpine vegetation using vegetation observations and topographic attributes. Landscape Ecology. (in press)

Sailor, D.J. & X. Li, 1999. A semi-empirical downscaling approach for predicting regional temperature impacts associated with climatic change. Journal of Climate 12, 103-114.

SPSS, 2002. SPSS for Windows. Release 10.0.5. http://www.spss.com

Ter Braak, C.J.F. & P. Smilauer, 1998. Canoco 4, Canoco Reference Manual and User's Guide to Canoco for Windows. Centre for Biometry, Wageningen, the Netherlands.

Van Dam, O., 2001. Forest filled with gaps: Effects of gap size on water and nutrient cycling in a tropical rain forest – A study in Guyana, Ph.D. Thesis, Faculty of Geographical Sciences, Utrecht. 208 pp.

Wesseling, C.G., D. Karssenberg, W.P.A. Van Deursen, & P.A. Burrough, 1996. Integrating dynamic environmental model in GIS: the development of a Dynamic Modelling Language. Transactions in GIS 1, 40-48.7

Foto: Saxifraga, Ben Delbaere

Up-scaling, interpolation and extrapolation of biogeochemical and ecological processes

Models
Predictability
Space
Time
Scale discrepanties

MARTIN WASSEN &
JOS VERHOEVEN

Prof. Dr. M.J. Wassen, Environmental Sciences, Faculty of Geography, Utrecht University, P.O.Box 80.115, 3508 TC Utrecht, The Netherlands. m.wassen@geog.uu.nl
Prof. Dr. J.T.A. Verhoeven, Landscape Ecology, Geobiology, Faculty of Biology, Utrecht University, P.O.Box 800.84, 3508 TB Utrecht, The Netherlands. j.t.a.verhoeven@bio.uu.nl

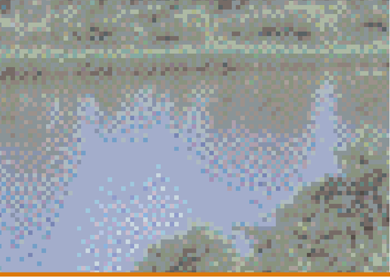
As regional and global scales become more important to ecologists, methods must be developed for the application of fine-scale knowledge to predict coarser-scale ecosystem properties. Scaling-techniques for aggregation, up-scaling, interpolation and extrapolation all have their specific constraints and possibilities. In this paper we address scale issues in ecological and landscape ecological research with special emphasis on up-scaling.
We conclude that in ecological modelling, limitations in data and their applicability for predictive modelling are more the rule than the exception, since collecting data on fine-grain patterns that are relevant at larger scales is generally costly and time consuming. Nevertheless, ecologically sound models can be obtained at the intermediate landscape scale (c. 100-10000 km^2) if they are based on a clear understanding of the scale at which relevant processes operate and serve as a template in choosing the appropriate scale in observation and modelling.

Although much progress has been made in understanding landscape processes, a thorough understanding of interactions between processes in and between landscape compartments and ecosystems is still largely lacking (Heymans *et al.*, 2002, Rietkerk *et al.*, 2002). This is partly due to discrepancies between the scales at which various processes operate, but more importantly, to discrepancies in scale regarding the questions asked, the models used and the data sources available (Gosselink & Lee, 1989). The scale of an investigation may have profound effects on the patterns one finds. Dynamic, statistical and spatial modelling are each used to integrate process information across scales. Such attempts have two directions. First, detailed studies carried out at finer scales can be integrated through dynamic models that can be used to study coarser scale processes. Typically, landscape models combine information on ecological processes with spatial information available through GIS (Arheimer & Brandt, 2000, Van den Bergh *et al.*, 2001, Pieterse *et al.*, 2002). A second approach to landscape analysis involves downscaling from studies that start at larger scales (e.g., entire river catchments) and work toward understanding relationships between geomorphology, geohydrology and land use patterns at smaller scales (see Burrough & Pfeffer, Whigham *et al.*, Mander *et al.*; this issue).

In this paper we analyse some scale issues in landscape science and we especially focus on up-scaling. After introducing some relevant definitions we address predictability in relation to space-time scaling. Next, we present three examples from the literature of scale-dependent processes each operating at a very different spatial and temporal scale. These examples are chosen to demonstrate that there are constraints in up-scaling approaches and they in fact show us that the problem of scale dependency is scale-independent. After discussing the implications of the scale of processes for data analysis and modelling we present two modelling studies: an empirical statistical model and a mechanistic model. In developing these models for up-scaling or aggregation we had to overcome several scale issues. Both approaches had their specific scale related constraints and possibilities, which may serve as general lessons. Finally, we formulate rules for application to avoid scaling errors.

Definitions

Generally speaking the scale of an object or process is its spatial or temporal dimension. In scaling studies the ability to detect patterns in space or time is a function of both the extent and the grain of an investigation (O'Neill *et al.*, 1986). *Extent* is defined generally as the overall area



encompassed by a study or the duration of the study. *Grain or support* is the size of the individual units of observation (Wiens, 1989) and is usually the largest area or time interval for which the property of interest is considered homogeneous (Bierkens *et al.*, 2000). *Coverage* is the ratio of the sum of areas or time intervals for all support units and the extent (Bierkens *et al.*, 2000). Thus, in a spatial example coverage refers to the part of the research area that is covered by samples, and in a temporal example it implies the sum of time intervals of observations divided by the total study time. Loosely speaking, up-scaling means transferring information from a smaller scale to a larger scale. More specifically *up-scaling* or *aggregation* is defined as increasing the support of the research area or the research time. Changing the extent of the research area or research time usually involves going from a smaller to a larger extent. Increasing the extent is called *extrapolation*. Interpolation involves increasing the coverage of the research area or research time, which is in fact the reverse of sampling (Bierkens *et al.*, 2000).

Note that MacArthur & Levins (1964) considered grain in a different way as we defined above. They defined grain as a function of how animals exploit resource patchiness in environments. The observational window of a consumer is then referred to as the grain at which a consumer perceives its habitat (O'Neill *et al.*, 1988, Milne, 1992, Ritchie, 1998). Differences in the scale of patchiness of the resource and the grain of observation by the consumer will affect the intensity of exploitation by the consumer. The size of the habitat that is covered by the consumer when searching for resource is then called the extent.

Predictability and space-time scaling

Our ability to predict ecological phenomena depends on the relationships between spatial and temporal scales of variation. Although there are no standard functions that define the appropriate units for space-time comparisons in ecology, with increased spatial scale, the time scale of important processes may also increase. This is because the relevant processes may operate at slower rates, their effects may involve time lags and their indirect effects may become increasingly important (Delcourt *et al.*, 1983, Clark, 1985). Thus, as the spatial scale of a system increases, so also may its temporal scale, although these space-time scalings differ for different systems. Studies over a long time and at a fine spatial scale have low predictive capacity at larger scales; they are simply too site-specific. Short-term studies conducted at broad spatial scales generally have a high apparent predictability but may be less capable of characterizing small-scale processes. This is pseudo-predictability since the natural dynamics of the system operate at much longer time scales than the period of study. It is as if we were to take two snapshots of a forest a few moments apart and use the first to predict the second (Wiens, 1989). The first photograph is a perfect predictor for the second, but it does not teach us anything about the relevant processes in a forest. Investigations that are designed to include a close correspondence between the time and space scales probably have the highest predictive power. In Fig. 1 we present a space-time diagram of ecological, hydrological and atmospheric processes illustrating the spatial and temporal scales that must be considered. Processes situated within the elliptic space are hypothesized to have a high predictability, whereas soil processes and peat growth are examples of processes with low predictability. Prediction of the activity of micro-decomposers or meteorological processes such as a thunderstorm event or the development of a cold front have a high apparent predictability over a wide range of scales.

In Figure 2 we depict the relationship between recovery time of events and scale (Dobson *et al.*, 1997). Remarkably,

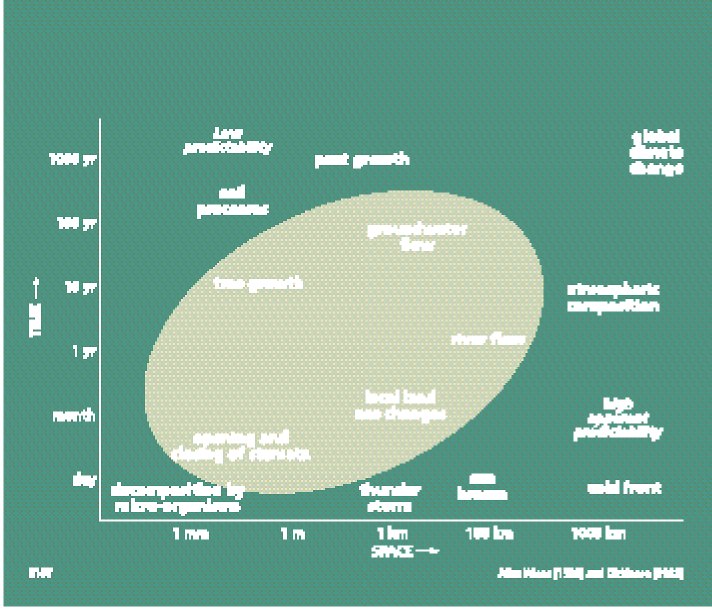


Figure 1. Predictability in relation to the space-time scaling of processes. (Left)

Figure 2. Recovery in relation to spatial scale. (Right)

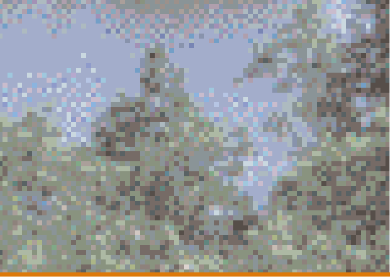
according to these authors a groundwater system needs a longer time to recover after groundwater exploitation than it takes for a part of the land surface to recover after an atomic bomb explosion.

An important implication from Figures 1 and 2 is that the questions asked by policy makers rarely are directed to the dynamics of the system and to the means (both financially and in time) that are given to those studying these processes. Often, ecologists have been urged by resource managers to answer questions and make and test predictions on relatively short time scales (some years), regardless of the spatial scale of the investigation. Politicians are frequently only interested in time horizons related to their careers, and since most of them are not in powerful positions before their mid forties, fifteen years ahead is about the maximum time span still enabling them to harvest within their active career. Thus, policy is often based on relatively short-term studies regardless the extent of the area and the rate at which the important processes occur. Especially, predicting the effects of human interference in processes such as peat growth, groundwater flow, groundwater composition and global climate processes require long term monitoring data. In comparison, short-term studies conducted at broad spatial scales have a high apparent predictability, since the natural dynamics of the system are so much longer than the period of study.

The difficulties in matching relevant scales in ecological modelling and the difficulties that they present in relating ecological information to policy decisions should be kept in mind when reading the three examples presented below. The examples illustrate that it is essential to identify the scale at which processes operate in order to design appropriate sampling schemes and perform sound analyses of data.

Example 1: Denitrification in floodplains

Denitrification is the process in which micro-organisms use oxygen obtained from nitrate for their respiration. The process results in the conversion of nitrate to gaseous forms of nitrogen (primarily N_2 and N_2O) that are lost to the atmosphere. Since denitrification decreases NO_3 concentrations and produces N_2O, the concentrations of NO_3 and N_2O in groundwater should be inversely related. The absence of this relationship found in field samplings (Weller *et al.* 1994) suggests that the N_2O pool is controlled by processes in addition to denitrification. N_2O can be produced by nitrification and can both be produced and consumed by denitrification. In addition, dissolved N_2O can be carried through the soil in groundwater or lost to the atmosphere. So, instead of measuring concentrations of two variables related to the process, it makes more sense to measure the rate of N_2O emission. This can be measured in closed chambers, in which



gasses emitted from the soil are measured. However closed chambers can only be used for short periods because temperature increase and gas buildup can change gas emission rates (Ryden & Rolston, 1983). Weller *et al.* (1994) used more than thirty chambers of 1x1 meter in a floodplain and did not find any obvious spatial pattern of N_2O emission rates nor any match with the pattern of N_2O or NO_3 in groundwater. Apart from N_2O emission rates being quite spatially variable, repeated measurements also showed big differences. Gas emission can also be measured using larger flow-through chambers. Larger chambers (20x1m) are more difficult to set up, but the constant flow of air minimizes temperature change and gas buildup over longer periods resulting in more useful data for monitoring emissions for days at a time (Jury *et al.*, 1982). Weller *et al.* (1994) installed two flow-through chambers in a floodplain, one on a low-lying, frequently waterlogged soil and one on a drier site. They observed a clear seasonal cycle with N_2O emission rates increasing from December to May and decreasing from September to December, paralleling seasonal temperature changes. They also observed diurnal variations in N_2O emission rates that correlated with temperature in the surface soil. The expected higher emissions in the low-lying floodplain site (having low redox status) were not observed, rather the reverse. Langeveld & Leffelaar (2002) modeled underground processes to explain N_2O profiles in the soil. Their model simulates several biological and physical processes. O_2 and CO_2 profiles were satisfactorily simulated indicating that the respiration rates used in their model were realistic. The N_2O profiles were less well simulated. They concluded that their assumption of homogeneity within soil layers was probably incorrect.

We conclude that it is hard to make realistic inferences about denitrification based on measurements that have high spatial and temporal variability. This is because it is a complex process operating on a fine scale in an environment where spatial heterogeneity of the factors influencing the process is large. This makes denitrification a difficult process to scale-up, to extrapolate and to model. Therefore generally valid estimates of NO_3 removal from groundwater by denitrification are lacking. An approach that might work for processes like denitrification is the search for so-called hot spots and hot moments, where the process is operating at a high rate (McClain *et al.*, 2003). These spots and moments probably cause the bulk of the nitrate removal in landscapes. They occur because at some points in space and time, an environmental factor that had limited the process is optimised. Denitrification requires low redox, pH>4, nitrate availability, carbon availability and a temperature higher than a critical minimum. Searching the conditions creating high rates in spatial data bases may help to identify such hot spots and moments.

Example 2: Biodiversity in ponds

Chase & Leibold (2002) tested Grime's (1979) hypothesis that local-scale species diversity first increases with slight increases of productivity, but then declines to low diversity when productivity is high. This so-called hump-shaped curve of species richness in response to productivity is supported by a wide variety of data and predictions of ecological models. This pattern is often seen in empirical studies at relatively small spatial scales (Waide *et al.*, 1999, Mittelbach *et al.*, 2001, Leibold, 1999, Dodson *et al.*, 2000). However, at regional spatial scales, species diversity often monotonically increases with increasing productivity instead of being hump-shaped (Curry & Paquin, 1987, Mittelbach *et al.*, 2001). Because studies performed at different spatial scales often consider different ecosystems and employ different methodology, it remains unclear if these relationships are scale-dependent or whether a single relationship holds across scales.

Chase & Leibold (2002) chose thirty ponds nested within ten watersheds. Each watershed had three ponds that were similar in productivity and total area. Local species richness within ponds was defined as the number of species in a pond, regional species richness as the total number of species observed in the three ponds within each watershed. At the local scale, both producer and animal species richness had a statistically significant hump-shaped relationship with primary productivity. In contrast, at the regional scale (among watersheds), species diversity linearly increased with productivity. An explanation might be that the differences in species composition among localities within regions increase with productivity. To test this hypothesis the authors calculated species dissimilarity of each watershed by quantifying the species compositional differences among the three ponds within a watershed. Species dissimilarity indeed increased with productivity; ponds within watersheds of low productivity shared the majority of their species, whereas ponds within watersheds of high productivity shared few.

Without going into the mechanisms causing these differences we may conclude that spatial scale dictates the productivity-diversity relationship. Species diversity, when viewed at different spatial scales, can respond in fundamentally different ways to the same environmental factor (productivity in the case of the ponds). Thus, straightforward up-scaling from local to regional scale is not appropriate in biodiversity studies.

Example 3: Variability in the feeding success of Sperm whales

Sperm whales (*Physeter macrocephalus*) feed on octopuses in the deep ocean at depths of 200-1000 meter. Large animals with a low reproductive rate and low mortality like the Sperm whale cannot react to environmental variation through changes in reproduction or mortality, thus they must possess mechanisms for surviving and averaging environmental variation over temporal scales less than their lifetimes and spatial scales less than their home-ranges. Whales come to the surface regularly to breath. When they dive again, their tail, the so-called fluke, is raised into the air. It is their habit to defecate at this particular moment, visible by a brown patch in the water. So the defecation rate is easy to observe and is defined as the proportion of fluke-ups at which the whale defecates.

Whitehead (1996) followed groups of Sperm whales in the Pacific and used temporal and spatial variation in defecation rates, which is a variation in feeding success, for assessing variation in octopus distribution in the deep ocean and the response of whales to this variation on a temporal and spatial scale. Mean defecation rates (per fluke-up), varied among years. When defecation rate is high (a high feeding success), the whales travel only short distances. If the variation in defecation rate is compared with the mean defecation rate, it appears that for time intervals of one day the coefficient of variation is somewhat less than the mean. For time intervals between 10 and 100 days variance is low and for intervals of years the variance is high compared to the mean.

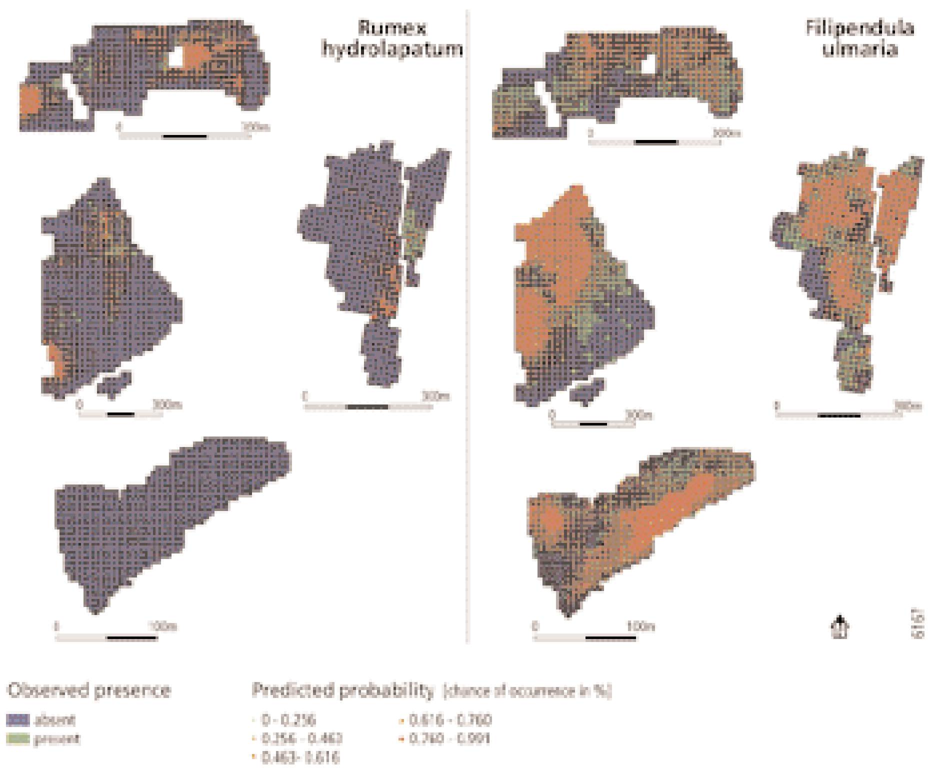
Apparently, temporal variability in the deep ocean is dominated by features with wavelengths of years. If we look at differences in variance with distance, we see that the variance over distances of about 100 kilometers is the same as that over periods of few days: somewhat less than the mean. However, over several hundred kilometres the variance in feeding success is larger, and similar to that over time periods of several years. Over larger distances it is about the same as the mean.

What can we learn from this study in which a proxy (defecation rate of Sperm whales) is used to estimate variability in octopus distribution and density in the deep ocean? Temporal variability in the deep ocean is governed by low-frequency, inter-annual features, just as was observed in studies focusing on variability at the surface (Steele 1985). These features are found in the Pacific in the California Current, the Humboldt Current (Peru) and the Equatorial Undercurrent influenced by El-Nino effects. Spatial coherence of such phenomena is limited to scales of a few hundred kilometres. The Sperm whales anticipate this by using migration over ranges of 300-1000 kilometers as their principal strategy for surviving in an unpredictable habitat. Migration thus allows Sperm whales to survive in an environment with unforeseen periods of food shortage. In other words, migration allows them to maintain high biomass and low reproductive rates in an environment, which at any location contains long unpredictable periods of food shortage.

Implications of the scale of processes for data analysis and modelling

The three examples of processes operating at very different spatial and temporal scales illustrate that scale does matter and that it is essential to identify the scale at which processes are operating. More specifically, one needs to identify the spatial scale at which the main factors operate or are distributed: the resources or variables influencing

Figure 3. Performance of the empirical statistical species response model VLITORS. For 38 species the models discriminated satisfactorily between areas but poorly within areas (shown is *Rumex hydrolapathum*). For 37 species the models discriminated satisfactorily between areas and within areas (shown is *Filipendula ulmaria*). For 10 species the models discriminated poorly between areas (not shown). Dots indicate the predicted probabilities; the background color of the grid cells indicate the observed presence of the species (blue absent, green present) (after De Becker *et al.*, 2001).



them (for example temperature, the availability of water or mineral nutrients, the distribution of plant cover or prey) and the organisms consuming a certain resource (for example denitrifying micro-organisms, herbivores or predators). It is also important to identify the spatial scale at which the interaction between resource and influencing variable or consumer takes place, e.g., N-sources in the soil and redox conditions; NO_3 and denitrifying micro-organisms; plant growth and herbivores; predator and prey. Van der Koppel *et al.* (in press) provide a simple framework that explains how differences in the spatial scale at which consumers and their resources function affect food chain theory. Such a framework is useful to identify critical scale aspects and to assess the risks of anthropogenic changes for trophic interactions by interfering with their functional scales.

Both the denitrification example and the Sperm whale example also illustrated that the temporal scale at which processes are influenced can vary a lot. Denitrification is affected by temperature and redox-conditions that vary during the day and also among seasons and years. The migration of Sperm whales varied among years. The study of biodiversity in ponds supported the notion that considerable insight can be gained by increasing the scale, both spatially and temporally, in which species diversity is viewed. Straightforward up-scaling from pond studies to catchments seems inappropriate in this case, since it would lead to erroneous conclusions for biodiversity in catchments, because of the non-linearity between the local scale and the catchment scale.

In the process of up-scaling among fine-scale components (such as biodiversity in local ponds) to predict coarser-scale properties of the aggregate (biodiversity in catchments), one has to be aware whether or not the relationship between variables and attributes is linear. If the model is linear it does not matter if the values of the variables are averaged before calculation of the average attribute value or if the average attribute value is obtained from averaging the separate calculated attribute values. If the relationship were non-linear such a procedure would result in an aggregation error (Rastetter *et al.*, 1992). Such an aggregation error will increase as the concavity of the non-linear function increases. To avoid such an error, when dealing with non-linear models, one has to calculate the attribute values first (apply the model at all grains, i.e., locations where input variables are known) and next average the function values (Bierkens *et al.*, 2000). Examples of such non-linear up-scaling functions are up-scaling from individual-leaf photosynthesis to full-canopy photosynthesis, up-scaling from small scale variation of the phreatic surface to regional models, or up-scaling of measured daily precipitation to average precipitation for a decade.

Scale problems in empirical statistical versus mechanistic modelling in landscape ecology

Ecological models generally link abiotic information (like water availability and quality) to organisms. Mechanistic ecological models, containing causal relationships derived from experimental studies, are available for relatively simple and thoroughly studied ecosystems (e.g., Van Liere and Gulati, 1992, Janse *et al.*, 1992). Mechanistic model development is both time-consuming and expensive. For the restoration of regional landscapes like watersheds and river valleys, generally applicable models valid for a range of ecosystems are required. These ecosystems and their interrelations are so complex that deterministic knowledge fully covering all processes is often not available and laborious experimental studies are not feasible. The two examples presented below serve as case studies illustrating the constraints related to scale issues in both types of modelling approaches. What we can learn

from these examples is that the general principle that discrepancies between the scale of observation, dominant processes, and model calculations should be avoided is frustrated in practice by limitations in data. Both modelling studies focus on river valleys: one empirical statistical approach focused on the response of plant species on changes in site factors (De Becker *et al.*, 2001, Bio *et al.*, 2002) and one a mechanistic approach focused on geochemical flows (Van der Peijl, 1997, Van der Peijl & Verhoeven, 1999, 2000).

Empirical model for plant species

This case is an example of spatial ecological predictive modelling, within the limitations imposed by data availability and model purpose given by environmental policy makers. Policy makers, e.g., water and nature managers, wanted a generally applicable model for Flemish river valleys although data only were available for four specific valleys. The data, collected from 1993-1997 in four nutrient-poor Flemish lowland river valleys, consisted of presence and absence records for groundwater-dependent plant species and abiotic site conditions describing management, soil, groundwater level and several groundwater chemistry parameters. Biotic data, management and soil were mapped in grids of adjacent regular square cells (20 x 20 m). Data on groundwater tables and water chemistry were collected at a limited number of point locations within each grid; hence, at a much smaller sampling scale (or support) and with extensive un-sampled surface in between. This example thus deals with a number of specific scaling constraints: limited extent of the study versus the need for a wider geographical applicability of the model; differences in support between variables; spatial autocorrelation.

The differences in support were relatively easy to overcome. The variables sampled with less support were spatially interpolated and up-scaled (to grid-cell size) to match the other data. This was done by block-kriging following a semi-variogram model, since this gave a much better result than standard block-kriging (De Becker *et al.*, 2001). Next, spatial auto-correlation in vegetation field records and model residuals was assessed through empirical semi-variograms; the residual semi-variograms indicated spatial structure not accounted for by the model's explanatory variables (cf. Albert & Mc Shane, 1995). Multiple logistic regression modelling was performed using two modelling frameworks. Generalized Linear Models - GLM- (Nelder & Wedderburn, 1972, McCullagh & Nelder, 1989) have been successfully applied in numerous ecological studies (e.g., Austin *et al.*, 1984, Margules *et al.*, 1987, Zimmermann & Kienast, 1999). Generalized Additive Models - GAM - (Hastie & Tibshirani, 1990, Yee & Mitchell, 1991) have been applied in more recent studies (e.g., De Swart *et al.*, 1994, Huntley *et al.*, 1995, Austin & Meyers, 1996, Bio *et al.*, 1998). Both enable ecologists to model species response to a wide range of environmental data using a link function (i.e., *logit*) between response and predictor variables. Generalized Additive Models form an extension of GLM. While GLM fit functions linear in their parameters, allowing for linear and polynomial response shapes, GAM are more flexible permitting both linear and complex additive response shapes, as well as a combination of the two within the same model (Hastie & Tibshirani, 1990). More than half of the species were modeled more accurately by GAM with data driven smooth response shapes instead of second-order polynomials. Model evaluation and comparison was based on cross-validation and model discrimination (Bio *et al.*, 2002). A factor coding for the four sampled valleys was most of the times very significant when added to the final regression model. This points at regional differences (between the valleys) in species distribution that are not ex-

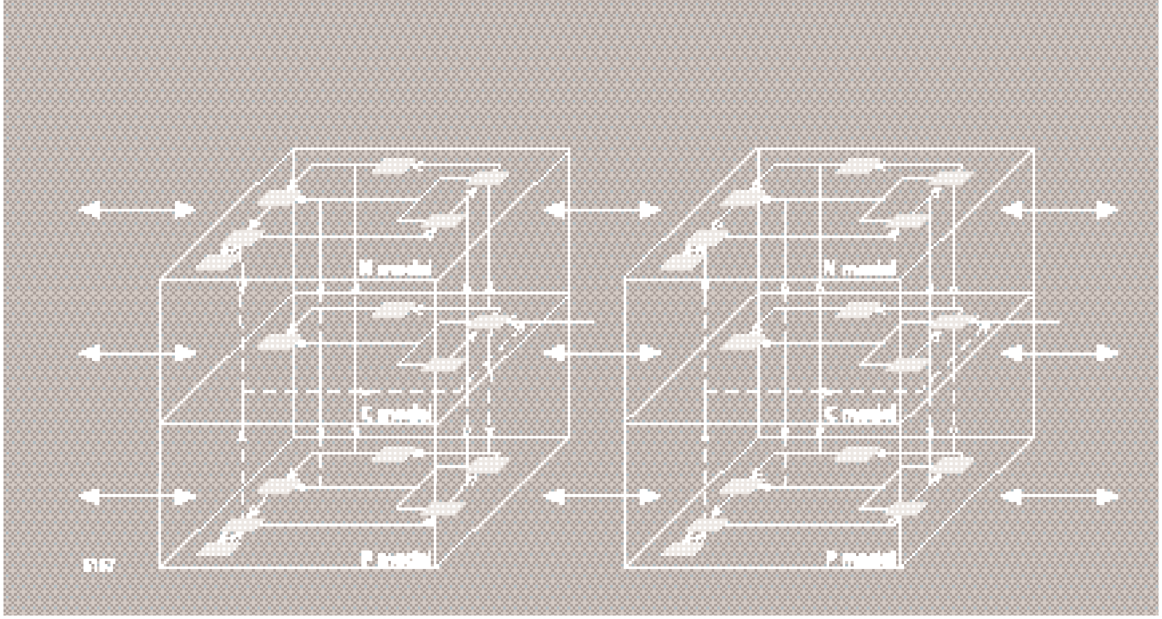


Figure 4. Conceptual diagram of a site-model consisting of two unit-models. Each unit-model consists of a nitrogen sub-model, a carbon sub-model and a phosphorus sub-model. Within these sub-models there is internal cycling. Landscape geochemical flows are shown between the unit-models (after Van der Peijl & Verhoeven, 2000).

plained by the models. There may be differences in species response to the explanatory variables due to valley-specific pseudo-correlations with non-modeled variables.

Overall, the regression models seemed ecologically sound and predicted species distribution in Flemish river valleys adequately, despite discrepancies between data quality and model assumptions. Figure 3 shows two examples illustrating model performance. The model of *Rumex hydrolapathum* only predicted well between areas and not within. The model for *Filipendula ulmaria* predicted observed distribution well both within and between areas. This study demonstrated that predictive modelling using standard statistical regression procedures can be reasonably successful with GLM or GAM in the presence of data with the following characteristics: non-homogeneous aggregated data; data that are spatially auto-correlated; partly interpolated and partly measured explanatory variables; explanatory variables and response variables collected at different scales; and correlated explanatory variables. However model application and inference should be handled with care, as assumptions of independent, error-free explanatory variables and independent errors are clearly not met. We observe that, in practice, models have to suit model purpose as well as possible even if data do not fully support model assumptions. Shortcomings, if not removable, should be assessed and, at least, communicated to the final user, just as model applicability and credibility. The models presented are, for instance, valid for nutrient poor river valleys only, as model input data do not include nutrient rich situations. So far, the predictive power of these models could not be examined on other regions. Validation against data collected elsewhere - i.e., an extrapolation in space - is a next step to be taken to see how far the applicability of these empirical models reaches (Bio *et al.*, 2002).

Mechanistic model for biogeochemical flows in wetland ecosystems

An example of a model describing carbon, nitrogen and phosphorus dynamics at the ecosystem level is the one developed by Van der Peijl & Verhoeven (1999) for river marginal wetlands. This model was developed in the framework of a European project on Functional Assessment in European Wetland Ecosystems (Maltby *et al.*, 1996) to analyse nutrient-related processes and their importance for ecosystem functions. In this case the constraints are: choices to be made in spatial and in temporal extent of the study in relation to the needed general applicability of the model and limited extrapolation possibilities.

The model is a dynamic simulation model in STELLA and has three layers, one for each element under investigation, i.e., carbon, nitrogen and phosphorus (Figure 4).

Each layer has a basically similar set-up with a number of plant and soil compartments with mass flows between them. Carbon fixation, nutrient uptake, grazing by large herbivores, decomposition, mineralization and denitrification are important processes described in the model. One of the main features of the model is a factor associated with soil redox potential, water table and soil oxygen content, which influences most process rates. The most important connections between the three model layers are the control of carbon fixation by nitrogen and phosphorus availability, and the control of mineralization by the litter C:N and C:P ratios.

The purpose of the model was to investigate the nature of the interactions between the C, N and P cycles, to assess what consequences these interactions have for water quality flowing through the wetland, for carbon sequestration and for greenhouse gas emissions. Further, attempts were made to quantitatively assess nutrient-related functions in river marginal wetlands and to simulate the effects of management and other human influences in (or outside) the wetland on these functions.

After the initial calibration and validation of the model with data collected in river marginal wetlands in England (Van Oorschot *et al.*, 1997), the model was used to test the nutrient transfers between two connected ecosystems, i.e., a wet, groundwater-fed slope and a floodplain along the river Torridge, SW England (Van der Peijl & Verhoeven, 2000). The hydro-geomorphic unit (HGMU) concept was used for defining a separate, complete unit-model for each of the two HGMU units within the wetland (Figure 4). These unit-models were connected by defining the flows of nitrogen and phosphorus between them. These flows, also called landscape geochemical flows, usually consist of flows of water containing N and P. The two units at the study site, Kismeldon Meadows, slope and floodplain, were separated by a ditch, which caught most of the run off and shallow groundwater flows from the slope. Only an estimated 1% of the N and P that left the slope unit in the water outflow reached the floodplain unit; the rest was caught in the ditch, which prevented the geochemical flows from taking their natural course. To examine the influence of this ditch, the model was run for the same site, but without the ditch. This is comparable to a situation of a restored site, where run-off and shallow groundwater containing nutrients can freely flow from the slope to the floodplain.

The computer simulation experiment reconnecting the slope and floodplain showed that this (1) increased the nutrient input into the floodplain, causing a higher biomass production, and (2) increased the wetness of the floodplain, causing slower decomposition, which together (3) led to a faster soil organic matter accumulation in the floodplain. Nutrient inflows became relatively more important compared to atmospheric deposition, especially for phosphorus. By connecting the slope and the floodplain, 20 % more nitrogen and 18% less phosphorus flowed into the river.

This model has a great level of detail with respect to the various biogeochemical processes involved and requires the availability of field data such as C, N and P stores in plants, soil organic matter, and other soil pools. It also requires many environmental parameters, such as climatic data, soil characteristics, water level fluctuations, etc. It has been shown to be effective in describing C-N-P interactions in wetland ecosystems, and has been sufficiently robust to implement a two-unit model in a landscape with two hydrologically connected wetland ecosystems (Van der Peijl & Verhoeven, 2000). Further spatial expansion of the model would be possible, although there is not much opportunity for modelling small-scale hydrological patterns in multi-unit (or grid-based) approaches.

Discussion

Empirical ecological models are often based on available data that were not explicitly collected for that purpose or on limited data sets especially collected for the purpose of model development (see De La Ville *et al.*, 1997, Ertsen *et al.*, 1998, Bio, 2000). Therefore, quantity and quality of data is of utmost importance. An ideal data set for ecological modelling contains a sufficient number of samples that are representative of and well distributed in the modeled geographical and environmental ranges, and that satisfy model assumptions. Unfortunately, such ideal data sets are rarely found, and the urgent need for swift restoration measures presses modelers to do with less than ideal data (see Olde Venterink & Wassen, 1997).

Classical statistical inference is based on the assumption of independent observations collected at randomly chosen locations (De Gruijter & Ter Braak, 1990). However, records of spatial dependence in ecological data are numerous (e.g., Rossi *et al.*, 1992; Tilman, 1994, Fielding & Bell, 1997), as neighboring samples tend to be more similar than samples further apart. Using standard statistics, the presence of spatial autocorrelation in data and in model residuals may render error estimates and associated significance tests unreliable. It may also affect model choice, as variable selection is generally based on explained and residual variance. Nonetheless, these data are generally treated as independent, random samples and modeled using classical statistical procedures (e.g., Nicholls, 1989, Hill, 1991, Buckland & Elston, 1993).

Recently, methods have been developed for the modelling of spatial dependence, or auto-correlation, in regression using, for instance, neighborhood information (Sokal & Oden, 1978a, b, Smith, 1994, Wu & Huffer, 1997). Geostatistical modelling of residual spatial dependence is an alternative approach under development (Pebesma *et al.*, 2000). However, for prediction at other sites or in different conditions, the use of spatial autocorrelation as model term or residual information has serious drawbacks. On the one hand, neighborhood or other spatial dependence information is not directly available, and the assumption that levels of spatial dependence for new sites or conditions are similar to those found at the modeled sites may not be valid. On the other hand, a spatial dependence term in the model will act as an indirect variable accounting for—and, possibly, masking part of—the effect of several direct, ecologically relevant variables. Vegetation records and records of abiotic site conditions tend to be auto-correlated too, and an explanatory variable defining the neighborhood of a site in terms of a species' occurrence will combine biotic (e.g., species' dispersal ability or inter-species competition) and abiotic (favorable or non-favorable site conditions) information. This will render robust but less informative and, possibly, less generalizable models. Only part of the spatial autocorrelation in the response variable is likely to be explained by the explanatory variables in the regression model. Assessment of the residual spatial variance can aid model evaluation, and highlight shortcomings in explanatory variables or model structure (e.g. Robertson & Freckman, 1995, Begg & Reid, 1997, Gotway & Stroup, 1997, Köhl & Gertner, 1997, Bio *et al.* 2003).

The main problem with empirical statistical species models is that there is little cause-effect knowledge incorporated. Of course, the choice of certain site conditions as potential predictor variables is based on knowledge of how these conditions affect species, but for the rest the model is merely statistic. The potential danger of pseudo-predictions is larger when less predictor variables are included, when the model is spatially extrapolated and especially when the short time scale of a study is not balanced to its large spatial scale. Van der Rijt *et al.* (1996) developed a model for predicting vegetation zonation in dependence of flooding in outer dike areas. They coupled several maps in a GIS and incorporated vegetation response regression models (based on a geographically small area) to these spatial data. The model was used for evaluation of the effects of different sluice management schemes on outer dike vegetation zonation in a wider area. There is nothing wrong with such predictions as long as flooding frequency and duration are the causal factors for vegetation zonation in all areas where the model is applied (Wassen *et al.*, 2003). The fact that we have to be cautious with extrapolation in time with this category of models is ironic, since this is what these models were developed for: extrapolations into the future.

High-detail (in terms of many processes incorporated) dynamic simulation models such as the one developed by Van der Peijl & Verhoeven (1999, 2000) have the advantage of integrating a strong knowledge base on biogeochemical interactions in order to analyze or predict the effects of major environmental drivers such as water level fluctuations and nutrient inputs in run-off on overall ecosystem performance, such as the water quality improvement function in wetlands. The drawback of the approach is that large data sets of site conditions are needed to implement the model. These would normally only be available if the site would have been intensively studied. Another limitation of the model is the coarse grain of study - it assumes homogeneous site conditions within certain hydrogeomorphic units. Such units subdivide the landscape in a discrete way, comparable with the 'ecotope' concept. Coarse-scale spatial variation in terms of multi-unit wetland landscapes can be tackled by running the model in every unit separately and using extra algorithms to describe the hydrological connections between the units. The model would be easier to apply if it would be simplified and implemented in a raster-GIS. There have been some first attempts to do this, and much simpler dynamic models simulating C-N-P interactions have been generated, which still kept their original level of predictability. If coarse-scale data for other units are unavailable, a statistical description of the fine-scale components across the extent of the coarser scale should be acquired. The fine-scale attributes can then be ranked by their contribution to the aggregation error. In such a way the important sources of error can be detected (Rastetter *et al.*, 1992). To detect scale-dependent processes and patterns, one depends on observation sets or model calculations of fine grain and large extent. Collecting data of fine grain and large extent is costly and time consuming. Therefore, an *a priori* choice of a certain scale of observation and/or modelling is often unavoidable. Clear understanding about the scale at which relevant processes operate is essential when choosing the appropriate scale of observation and modelling. A general guideline in choosing an appropriate scale of study is that discrepancies between the scale of observation, dominant processes, and model calculations should be avoided (Rietkerk *et al.*, 2002). Since in most environmental studies such discrepancies are a given and thus cannot be avoided, they should be explicitly acknowledged.

Although we have identified a whole range of pitfalls and possible sources of error involved in attempts to scale up patterns and processes from small-scaled site studies, we can identify several promising approaches, which can be further developed. A first approach is the use of statistical regression of spatial data, with attention for spatial autocorrelation including assessment of spatial variance. It is important that statistical correlations found with these models are validated with knowledge on cause-effect relations. If such knowledge does not exist for the specific relations found, these should be interpreted with care and should ideally still be studied in a causal-analytical way. A second approach is the implementation of simplified mechanistic models of biogeochemical and population ecological processes in a raster-GIS, with simultaneous modelling of the spatial relationships between raster cells in a hydrological model. The mechanistic model should be parameterised and calibrated with data from studies in one or two spatial cells in the study area. Only a limited number of sensitive parameters for the model have to be measured in all the raster cells.

We advocate a combination of approaches, empirical models for species response and mechanistic modelling of biogeochemical processes, in order to gain insight into regional landscapes and to allow for some form of prediction of environmental and management effects on these systems. Van den Bergh *et al.* (2001), Pieterse *et al.* (2002) and Gielczewski (2003) provide good examples of attempts of such integrated models. Although these models also suffer from scale discrepancies, they at least provide an explicit framework revealing them, since the questions asked have to be translated into spatial scenarios and subsequently into input maps whereas the models provide output maps and for all of these steps the spatial and temporal scale is clear.

Acknowledgement

We thank Dennis Whigham and an anonymous referee for their helpful suggestions.

Abstract

Inquiries into the issue of scale become increasingly important in the field of landscape ecology and natural resource modelling and analysis. Scales of observation and modelling are often pre-set based on the *a priori* description of the system of study. In this paper we focus on upscaling approaches. We emphasize that predictability depends on the relation between the spatial and the temporal scale of study. Three examples of scale dependent processes illustrate the importance of identifying the scale at which processes operate to avoid erroneous conclusions. Two modelling studies show a number of scale related bottlenecks in data, interpolation, extrapolation and modelling. In statistical modelling of spatial data spatial dependence should be examined, truly independent validation data sets should be available and spatial extrapolation should be done with care. In mechanistic modelling of processes spatial up-scaling requires information on landscape heterogeneity and how this influences the modelled processes. Although a general guideline in choosing an appropriate scale of study is that discrepancies between the scale of observation, dominant processes and model calculations should be avoided, in most landscape ecological studies such discrepancies are a given. They should be explicitly acknowledged and the information in this paper may help in recognizing them and dealing with them.

References

Albert P.S. & L.M. McShane. 1995. A generalized estimating equations approach for spatially correlated binary data: applications to the analysis of neuro-imaging data. Biometrics 51: 627-638.

Arheimer, B. & M. Brandt. 2000. Watershed modelling of nonpoint nitrogen losses form arable land to the swedish coast in 1985 and 1994. Ecological engineering 14: 389-404.

Austin M.P., R.B. Cunningham & P.M. Fleming 1984. New approaches to direct gradient analysis using environmental scalars and statistical curve-fitting procedures. Vegetatio 55:,11-27.

Austin M.P. & J.A. Meyers. 1996. Current approaches to modelling the environmental niche of eucalypts: implication for management of forest biodiversity. Forest Ecology and Management 85: 95-106.

Begg G.S. & J.B. Reid. 1997. Spatial variation in seabird density at a shallow sea tidal mixing front in the Irish Sea. ICES Journal of Marine Science 54: 552-565.

Bierkens, M.F.P., P.A. Finke & P. de Willigen. 2000. Upscaling and downscaling methods for environmental research. Kluwer Academic Publishers. Dordrecht.

Bio A.M.F., R. Alkemade & A. Barendregt. 1998. Determining alternative models for vegetation response analysis: a non-parametric approach. Journal of Vegetation Science 9: 5-16.

Bio, A.M.F. 2000. Does vegetation suit our models? Data and model assumptions and the assessment of species distribution in space. Ph.D. Thesis, Utrecht University.

Bio, A.M.F., P. De Becker, E. De Bie, W. Huybrechts & M.J. Wassen. 2002. Prediction of plant species distribution in lowland river valleys in Belgium: modelling species response to site conditions. Biodiversity and Conservation 11: 2189-2216.

Buckland S.T. & D.A. Elston 1993. Empirical models for the spatial distribution of wildlife. Journal of Applied Ecology 30: 478-495.

Chase, J.M. & M.A. Leibold. 2002. Spatial scale dictates the productivity-biodiversity relationship. Nature 416: 427-430.

Clark, W.C. 1985. Scales of climate impacts. Climatic Change 7: 5-27.

Curry, D.J. & V. Paquin. 1987. Large-scale biogeographical patterns of species richness of trees. Nature 329: 326-327.

De Becker P., E. De Bie, W. Huybrechts, A.M.F. Bio & M.J. Wassen 2001. Ontwikkeling van een hydro-ecologisch model voor vallei-ecosystemen in Vlaanderen, VLITORS. Intituut voor Natuurbehoud, Brussels.

De Gruijter J.J. & C.J.J. Ter Braak. 1990. Model-free estimation from spatial samples: A reappraisal of classical sampling theory. Mathematical Geology 22: 407-415.

De La Ville N., S. Cousins & C. Bird. 1997. Habitat suitability analysis using logistic regression and GIS to outline potential areas for conservation of the Grey Wolf (Canis lupus). Presented at the Conference: 'GIS Research UK', University of Leeds, Leeds 9-11 April 1997.

De Swart E.O.A.M., A.G. Van der Valk, K.J. Koehler & A. Barendregt 1994. Experimental evaluation of realized niche models for predicting responses of plant species to a change in environmental conditions. Journal of Vegetation Science 5: 541-552.

Delcourt, H.R., P.A. Delcourt, & T. Webb. 1983. Dynamic plant ecology: the spectrum of vegetation change in space and time. Quartenary Science review 1: 153-175.

Dickinson, R.E. 1988. Atmospheric systems and global change. In: Scales and Global Change, pp. 57-80. T. Roswall, Woodmansee, R. & P.Risser (eds.), Wiley, New York.

Dobson, A.P., A.D. Bradshaw & J. Baker 1997. Restoring ecosystems – Response. Science 278: 999-1001.

Dodson, S.I, S.E. Arnott & K.L. Cottingham. 2000. The relationship in lake communities between primary productivity and species richness. Ecology 81: 2662-2679.

Ertsen A.C.D., A.M.F. Bio, W. Bleuten & M.J. Wassen. 1998. Comparison of the performance of species response models in several landscape units in the province of Noord-Holland, The Netherlands. Ecological Modelling 109: 213-223.

Fielding A.H. & J.F. Bell 1997. A review of methods for the assessment of prediction errors in conservation presence/absence models. Environmental Conservation 24: 38-49.

Gielczewski, M. 2003. An integrated modelling approach for sustainable management of the Narew River Basin (Poland). Ph.D.Thesis, Utrecht University.

Gosselink, J.G. & L.C. lee, 1989. Cumulative impact assessment in bottomland hardwood forests. Wetlands 9: 83-174.

Gotway C.A. & W.W. Stroup. 1997. A generalized linear model approach to spatial data analysis and prediction. Journal of Agricultural, Biological, and Environmental Statistics 2: 157-178.

Grime, J.P. 1979. Plant strategies and vegetation processes. Wiley and Sons, Chicester.

Hastie T.J. & R.J. Tibshirani. 1990. Generalized additive models. Chapman and Hall, London.

Heymans, J.J., R.E. Ulanowicz & C. Bondavalli. 2002. Network analysis of the South Florida Everglades graminoid marshes and comparison with nearby cypress ecosystems. Ecological Modelling 149: 5-24.

Hill M.O. 1991. Patterns of species distribution in Britain elucidated by canonical correspondence analysis. Journal of Biogeography 18: 247-255.

Huntley B., P.M. Berry, W. Cramer & A.P. McDonald. 1995. Modelling present and future ranges of some European higher plants using climate response surfaces. Journal of Biogeography 22: 967-1001.

Janse J.H., T. Aldenberg & P.R.G. Kramer. 1992. A mathematical model of the phosphorus cycle in Lake Loosdrecht and simulation of additional measures. Hydrobiologia 233: 119-136.

Jury, W.A., J. Letey, & T. Collins. 1982. Analysis of chamber methods used for measuring nitrous oxide production in the field. Soil. Science Society of America Journal 46: 250-256.

Köhl, M. & G. Gertner 1997. Geostatistics in evaluating forest damage surveys: considerations on methods for describing spatial distributions. Forest Ecology and Management 95: 131-140.

Langeveld, C.A. & P.A. Leffelaar. 2002. Modelling belowground processes to explain field-scale emissions of nitrous oxide. Ecological Modelling 149: 97-112.

Leibold, M.A. 1999. Biodiversity and nutrient enrichment in pond plankton communities. Evol. Ecol. Res. 1: 73-95.

MacArthur, R.H. & R. Levins. 1964. Competition, habitat selection and character displacement in a patchy environment. Proceedings of the National Academy of Science of the USA, 51: 1207-1210.

Maltby, E., D.V. Hogan, & R.J. McInnes. 1996. EUR 16132. Functional analysis of European wetland ecosystems - Phase 1 (FAEWE). European Commission, Office for Official Publications of the European Communities, Luxemburg, 1-448 pp.

Margules C.R., A.O. Nicholls & M.P. Austin 1987. Diversity of Eucalyptus species predicted by a multi-variable environmental gradient. Oecologia 71: 229-232.

McClain, M.E., E.W. Boyer, C.L. Dent, S.E. Gergel, N.B. Grimm, P.M. Groffman, S.C. Hart, J.W. Harvey, C.A. Johnston, E. Mayorga, W.H. McDowell & G. Pinay. 2003. Biogeochemical hot spots and hot moments at the interface of terrestrial and aquatic ecosystems. Ecosystems 6: 301-312.

McCullagh P. & J.A. Nelder. 1989. Generalized linear models. Second edition. Chapman and Hall, London.

Milne, B. T. 1992. Spatial Aggregation and Neutral Models in Fractal Landscapes. American Naturalist 139:32-57.

Mittelbach, G.G., P. Jones, & L.W. Emmerson. 2001. What is the observed relationship between productivity and diversity? Ecology 82: 2381-2396.

Nelder J.A. & R.W.N. Wedderburn. 1972. Generalized linear models. Journal of the Royal Statistical Society A 135: 370-384.

Nicholls, A.O. 1989. How to make biological surveys go further with generalised linear models. Biological Conservation 50: 51-75

Olde Venterink, H.G.M. & M.J. Wassen. 1997. A comparison of six models predicting vegetation response to hydrological habitat change. Ecological Modelling 101: 347-361.

O'Neill, R.V., DeAngelis, D.L., Waide, J.B. & T.F.H. Allen. 1986. A hierarchical concept of ecosystems. Princeton University Press, Princeton, New Jersey.

O'Neill, R. V., B. T. Milne, M. G. Turner & R. H. Gardner. 1988. Resource utilization scales and landscape pattern. Landscape Ecology 2:63-69.

Pebesma E.J., R.N.M. Duin & A.M.F. Bio 2000. Spatial interpolation of sea bird densities on the Dutch part of the North Sea. Department of Physical Geography, Faculty of Geographical Sciences, Utrecht University.

Pieterse, N.M., A.W.M. Verkroost, C. Kwakernaak, H. Olde Venterink & M.J. Wassen. 2002. A decision support system for restoration of stream valley ecosystems. Landscape Ecology 17: 69-81.

Rastetter, E.B., A.W. King, B.J. Cosby, G.M. Hornberger, R.V. O,Neill, & J.E. Hobbie. 1992. Aggregating fine-scale ecological knowledge to model coarser scale attributes of ecosystems. Ecological Applications 2(1): 55-70.

Rietkerk, M., J. Van de Koppel, L. Kumar, F. Van Langevelde & H.H.T. Prins (eds.). 2002. Natural resource modelling; the ecology of scale (editorial). Special Issue Ecological Modelling 149(1-2), 211 pp.

Rietkerk, M., M.C. Boerlijst, F. Van Langevelde, R. HilleRisLambers, J. van de Koppel, L. Kumar, H.H.T. Prins & A.M. de Roos. 2002. Self-organization of vegetation in arid ecosystems. The American Naturalist 160: 524-530.

Ritchie, M. E. 1998. Scale-dependent foraging and patch choice in fractal environments. Evolutionary Ecology 12:309-330.

Robertson G.P. & D.W. Freckman. 1995. The spatial distribution of nematode trophic groups across a cultivated ecosystem. Ecology 76: 1425-1432.

Rossi R.E., D.J. Mulla, A.G. Journel & E.H. Franz. 1992. Geostatistical tools for modelling and interpreting ecological spatial dependence. Ecological Monographs 62: 277-314.

Ryden, J.C. & D.E. Rolston, 1983. The measurement of denitrification. In: J.R. Freney & J.R. Simpson (eds). Gaseous loss of nitrogen from plant-soil systems. Dr. W. Junk Publishers, The Hague pp. 91-132.

Smith P.A. 1994. Autocorrelation in logistic regression modelling of species distributions. Global Ecology and Biogeography Letters 4: 47-61.

Sokal R.R. & N.L. Oden. 1978a. Spatial autocorrelation in biology. 1. Methodology. Biological Journal of the Linnean Society 10: 199-228.

Sokal R.R. & N.L. Oden. 1978b. Spatial autocorrelation in biology. 2. Some biological implications and four applications of evolutionary and ecological interest. Biological Journal of the Linnean Society 10: 229-249.

Steele, J.H. 1985. A comparison of terrestrial and marine ecological systems. Nature 313: 355-358.

Tilman D. 1994. Competition and biodiversity in spatially structured habitats. Ecology 75: 2-16.

Van den Bergh, J., Barendregt, A., Gilbert, A., Van Herwijnen, M., van Horssen, P., Kandelaaars, P. & C. Lorenz. 2001. Spatial economic –hydroecological modelling and evaluation of land use impacts in the Vecht wetlands area. Environmental Modelling and Assessment 6: 87-100.

Van de Koppel, J., R. Bardgett, J. Bengtsson, H. Olff, M. Rietkerk, C. Rodrigues-Barrueco, M.J. Wassen & V. Wolters. Scale effects on trophic interactions. Ecosystems (in press).

Van Liere L. & Gulati R.D. (eds) 1992. Restoration and recovery of eutrophic lake ecosystems in the Netherlands. Hydrobiologia 233: 283-287

Van der Peijl, M. 1997. Modelling of biogeochemical processes and spatial patterns in freshwater wetlands. Ph.D.Thesis, Utrecht University.

Van der Peijl, M. & J.T.A. Verhoeven. 1999. A model of carbon, nitrogen and phosphorus dynamics and their interactions in river marginal wetlands. Ecological Modelling 118: 95-130.

Van der Peijl, M. & J.T.A. Verhoeven. 2000. Carbon, nitrogen and phosphorus cycling in river marginal wetlands; a model examination of landscape geochemical flows. Biogeochemistry 50: 45-71.

Van der Rijt, C.W.C.J., L. Hazelhoff & C.W.P.M. Blom. 1996. Vegetation zonation in a former tidal area: A vegetation-type response model based on DCA and logistic regression using GIS. Journal of Vegetation Science 7: 505-518.

Van Oorschot, M.M.P., E. Robbemont, M. Boerstal, M. Van Strien & M. Kerkhoven-Schmitz. 1997. Effects of enhanced nutrient availability on plant and soil nutrient dynamics in two English riverine ecosystems. Journal of Ecology, 85:167-179

Waide, R.B., M.R. Willig, C.F. Steiner, G. Mittelbach, L. Gough, S.I. Dodson, G.P. Juday, & R. Parmenter 1999. The relationship between primary productivity and species richness. Annu. Rev. Ecol. Syst 30: 257-300.

Wassen. M.J., W.H.M. Peeters, & H. Olde Venterink. 2003. Patterns in vegetation, hydrology and nutrient availability in an undisturbed river floodplain in Poland. Plant Ecology 165: 27-43.

Weller, D.E., D.L. Correll, & T.E. Jordan. 1994. Denitrification in riparian forests receiving agricultural discharges. In: W.J. Mitsch (ed). Global wetlands: Old world and new. Elsevier Science, Amsterdam.

Whitehead, H. 1996. Variation in the feeding success of Sperm whales: temporal scale, spatial scale and relationship to migrations. Journal of Animal Ecology 65(4): 429-438.

Wiens, J.A. 1989. Spatial scaling in ecology. Functional Ecology 3: 385-397.

Wu, H. & F.W. Huffer 1997. Modelling the distribution of plant species using the autologistic regression model. Environmental and Ecological Statistics 4: 49-64.

Yee, T.W. & N.D. Mitchell 1991. Generalized additive models in plant ecology. Journal of Vegetation Science 2: 587-602.

Zimmermann, N.E. & F. Kienast 1999. Predictive mapping of alpine grasslands in Switzerland: species versus community approach. Journal of Vegetation Science 10: 469-482.

Flexibility in landscape ecology

Much work remains if we are to successfully apply emerging principles from the field of landscape ecology to meeting the needs of human societies around the world. Humans, for example, are having an enormous impact on marine and estuarine ecosystems and most problems result from a lack of understanding of how marine ecosystems function combined with a political unwillingness to apply ecological principles to management. The continued decline in the ecological health of global ecosystems will eventually result in ecological and human disasters and few possibilities of meaningful restoration.

We have already reached the point where the costs of large-scale restoration efforts are beyond the means of many countries. Three examples from the US indicate the scale of the problem. Hundreds of millions of dollars have been spent to restore the Chesapeake Bay, the largest estuary in the U.S., but efforts have only resulted in slowing the rate of decline of most natural resources and the Bay still faces an uncertain future. Several billions of dollars are currently being allocated to restore the Florida Everglades; yet most ecologists believe that the restoration plans, developed mostly from an engineering perspective, have little chance of success because they do not adequately integrate principles of ecosystem and landscape ecology. The Mississippi is the largest river in the U.S. and recent analyses have shown that the long-term effects of human activities in the catchment and in estuarine and marine areas have resulted in environmental problems in the Gulf of Mexico that are much greater than those in the Chesapeake Bay or Florida. Plans to restore only part of the Mississippi system, wetlands along the coast of Louisiana, is estimated to cost $20 billion or more and the plan is based on a weak understanding of ecological functioning at the landscape and ecosystem scales.

Even in relatively pristine areas, a lack of knowledge about how landscapes and ecosystems function hinders planning. In the Kenai Peninsula of Alaska, for example, the Anchor River catchment supports valuable salmonid populations. Development in the catchment has had little impact on salmon to date but small-scale disturbances are beginning to increase in frequency. What scale of disturbances will result in significant impacts on salmon in the Anchor River catchment? We currently can not answer that question because we know little about linkages between ecosystems in the watershed. Subsequently, it is difficult to convince the public that the level of current and projected future activities may have negative impacts on salmon, an extremely valuable local economic resource.

What do these examples mean for scientists involved in the field of landscape ecology? First, it means that we need to work harder and faster to develop the field. The level of resource degradation is already high and the rate at which landscapes are being degraded is certainly increasing. The examples suggest that efforts to understand how landscapes function needs to move forward at several scales. In developed countries, an increased understanding of landscape dynamics is required to develop ecologically useful management and restoration plans. Management plans in these instances are likely to be complex and restoration efforts costly. In developed and developing countries where landscapes are currently in relatively pristine condition, landscape studies should focus on identifying key indicators of ecosystem and landscape health. Monitoring of key indicators can be used to identify when and where elements of catchments are beginning to be impacted by human activities. In developing countries, resource limitations will limit most management and restoration efforts to small-scale community based initiatives that require a clear understanding of landscape interactions followed by identification of key indicators of resources and processes that can be monitored to indicate where and when problems occur. Education and community involvement, however, will likely be the key to successful application of landscape principles, no matter the scale of the study and the cost of management or restoration. We can study landscapes and ecosystems forever but our efforts will be minimal unless we are able to communicate our knowledge to individuals, communities, and governments. The task is enormous. A recent commentary in *Frontiers in Ecology and the Environment* indicated that The Netherlands is the only developed country in the world with a national policy of managing its environment for purposes of restoring the ecological health and integrity of its lands. The remainder of the world awaits the results of our efforts.

DENNIS F. WHIGHAM

Prof. dr. D.F. Whigham occupies the WLO chair on landscape ecology at the Section of Landscape Ecology, Department of Geobiology, Utrecht University. He is also attached to the Smithsonian Environmental Research Center, Edgewater, MD (USA).


Foto: Saxifraga, Ben Delbaere

Handling scales when estimating Swedish nitrogen contribution from various sources to the Baltic Sea

Nitrogen
Hydrology
Modelling
Catchment
Scales

At the national and international policy level, there is an increasing demand for overall estimations of the contribution of the runoff from large regions or whole countries to the nutrient loadings of river basins and coastal areas. This article decribes a methodology involving scaling up data on nitrogen leaching and transport from the site scale to the scale of river basins and, eventually to the scale of Sweden as a whole. The upscaling methods are based on the linkage of leaching and transport models at the site scale with a nested model system involving regional hydrological models and source apportionment of N loadings towards the Baltic sea.

BERIT ARHEIMER

Dr. B. Arheimer. Swedish Meteorological and Hydrological Institute (SMHI), SE-601 76 Norrköping, Sweden. berit.arheimer@smhi.se

The nutrient loads to the Baltic Sea have increased successively during the 20th century (Larsson *et al.*, 1985) and have resulted in an ongoing degradation of the environment (Cloern, 2001). These negative effects have taken such proportions that the riparian countries were forced to take remedy actions. One obvious strategy is to reduce the nutrient load from land to sea, and most countries have reduced their point sources by 50% for phosphorus (P). However, this goal has not been achieved for the largest point sources, which are situated in Poland and Russia (Lääne *et al.*, 2002). Nitrogen (N) reduction from point sources, as well as the overall reduction of load from diffuse sources, has in most countries been less successful. Recent estimates based on official statistics indicate that load from agriculture constitutes approximately 60% of the anthropogenic N load and more than 25% of the anthropogenic P load to the Baltic Sea (Lääne *et al.*, 2002). The largest reduction achieved for arable leaching is mainly related to the economic breakdown of the agricultural sector in the transition countries. So far it has been difficult to monitor the effects, which is mainly due to large storage of nutrients in the soil and water systems (Stålnacke *et al.*, 2002). The nations around the Baltic Sea regularly report their national load to the Helsinki Commission (HELCOM), and for the latest pollution load compilation it was also obliged to specify the contribution from various sources.

Water management in Sweden is going through dramatic changes at present, related to the adoption of the EU Water Framework Directive, a new Environmental Code and revised Environmental Quality Objectives. New policies including catchment-based management plans have been suggested, which also demand catchment-based knowledge of nutrient transport processes and appropriate tools for landscape planning. Although Sweden has effectively reduced the nutrient load from treatment plants and industries during the past decades, the problem of eutrophication is not yet solved due to nutrient leaching from diffuse sources, such as arable land, rural households, and traffic. These sources are difficult to monitor and models must be applied to quantify their load, and to quantify possible load reductions, which have been or will be achieved in management programs (Figure 1).

A catchment model for the national scale (HBV-N) has therefore been developed to be used both for international reporting and for scenario estimates for more efficient control strategies. This paper provides an example of an interdisciplinary methodology that focuses on water quality and management issues at different scales (Figure 2). It includes upscaling of leaching models from the site scale to whole river basins in order to enable estimation of the N loading from the entire country with relatively high spatial and temporal resolution. The paper mainly describes how the transfers between scales have been handled and gives some model results from the application of the model concept for the whole country of Sweden (about 450 000 km^2).

Figure 1. Several reasons why dynamic and predictive models are useful tools in environmental assessment, management planning, in the implementation process of measures, and to follow-up environmental goals (exemplified with the structure of the catchment model HBV).

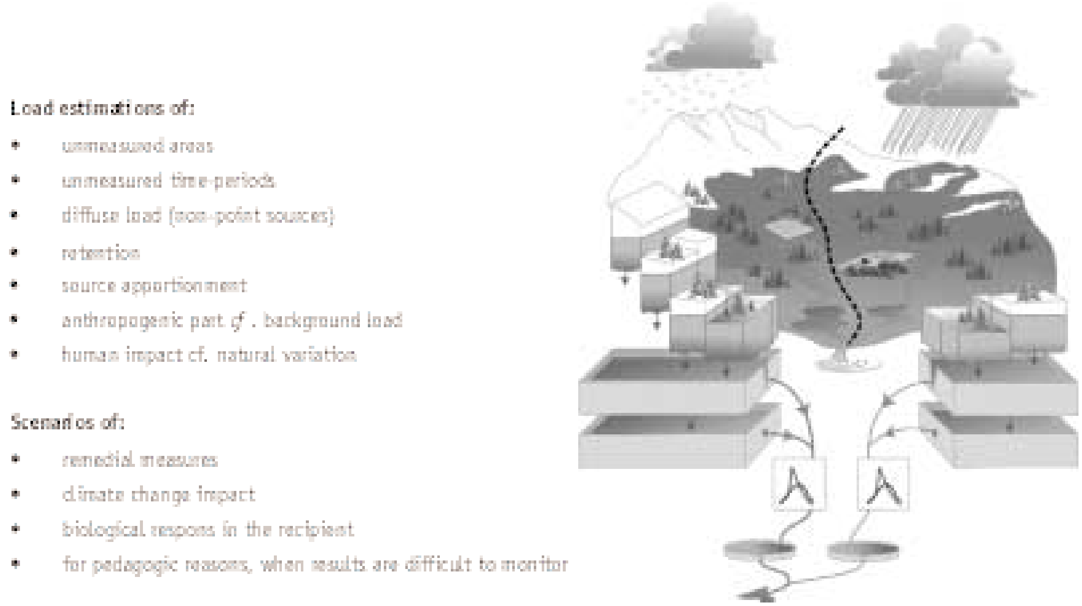
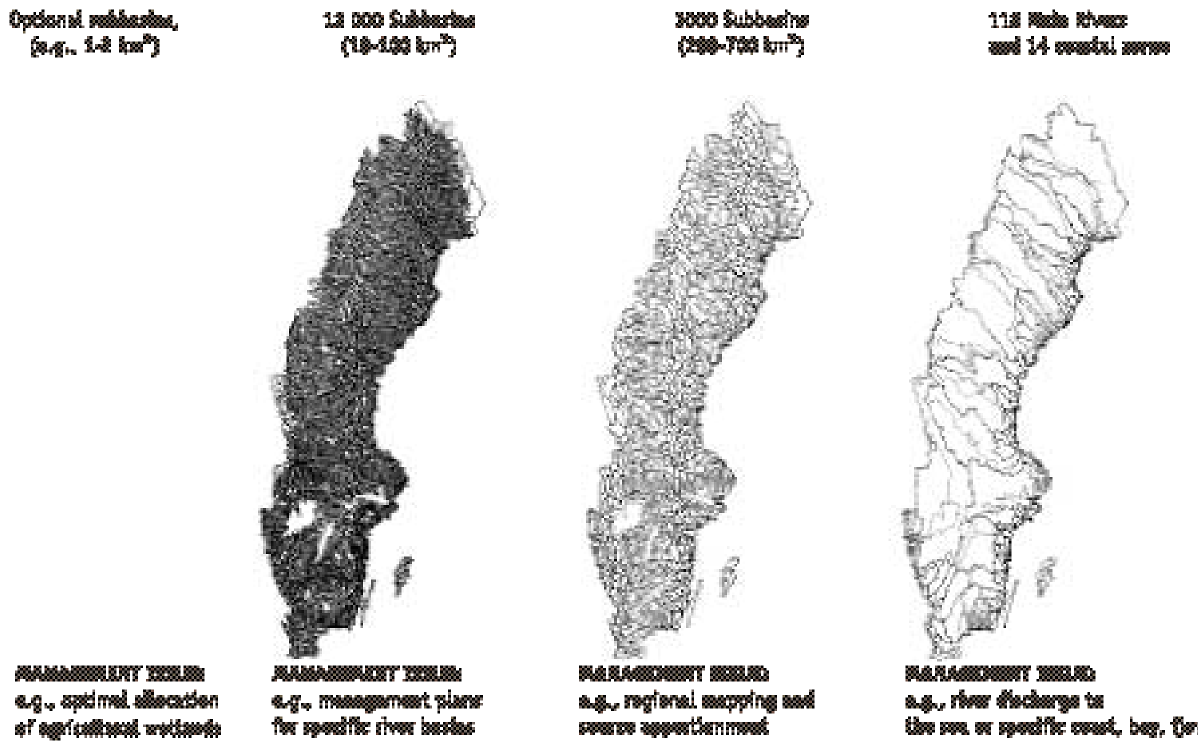


Figure 2. Various scales of catchment modelling with HBV-N in Sweden, using different databases for different management issues.



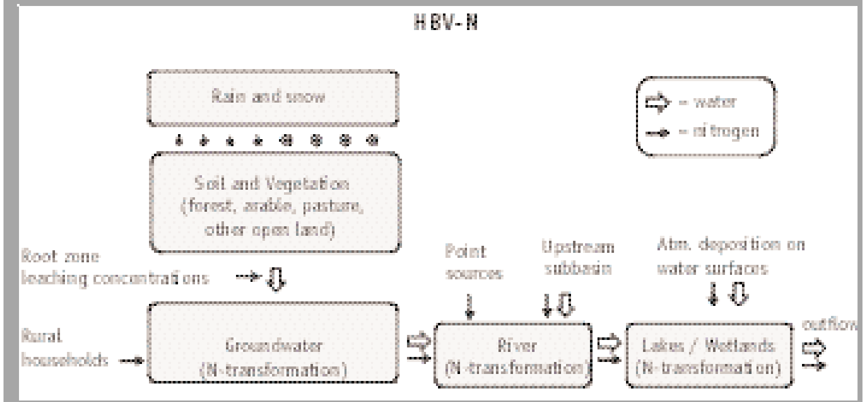


Figure 3. Schematic structure of the dynamic catchment model HBV-N.

Method

The catchment model HBV-N (Figure 3.) has been applied for the national scale within a nested model system, called TRK (Table 1), which calculates flow-normalised annual average of nutrient gross load, N retention and net transport, and source apportionment of the N load reaching the sea (Brandt and Ejhed, 2003). The TRK system consists of several submodels with different levels of process descriptions that are linked together (Bergstrand *et al.*, 2002). Dynamic and detailed models are included for arable leaching, water balance, and N removal. Daily simulations are made for a 20-year time-period. The results are subsequently aggregated over the entire 20-year period to cancel out short-term weather-induced variations. Landscape information, leaching rates and emissions are combined through GIS. N transport is simulated through the hydrological model, which accounts for transport and decay within subbasins, and routing through the river system, e.g., when passing lakes, towards the sea. During decay N removal may occur.

Up-scaling of root-zone leaching

Leaching concentrations from arable land is calculated with the physically based SOILN model (Johnsson *et al.*, 1987) for different field categories. General model input parameters are assumed to represent the average for a whole agricultural region, using the SOILNDB concept (Johnsson *et al.*, 2002). Sweden is then divided into 22 agricultural regions, based on climate and agricultural character. For each region separate calculations are made for 9 soil types, 13 crops, and 2 fertilisation strategies. A crop sequence generator is applied to obtain the average leaching concentration for all acceptable combinations in the crop rotation. Time-series of 20-30 years (calculated with a daily time-step) are used to consider weather-induced variability. Accumulation of the loads over the entire calculation period results in one aggregated concentration (i.e., not affected by temporal variations) for each combination of region, soil and crop. For each subbasin, an average root-zone concentration is then calculated based on land-use information of crop and soil distribution. This average leaching concentration is assigned to the water discharge from the root zone in the HBV-N catchment model (Figure 3).

Up-scaling of water balance and discharge

The water balance at the catchment-scale is estimated by using the conceptual rainfall-runoff model HBV (Bergström, 1995; Lindström *et al.*, 1997), which makes daily calculations in semi-lumped subbasins that are coupled along the river network. The HBV model consists of routines for snow melt and accumulation, soil moisture, runoff response and routing through lakes and streams. The runoff generation routine is the response function, which transforms excess water from the soil moisture zone to runoff. It also includes the effect of direct precip-

Table 1. Definition of spatial and temporal scales in the national model application within TRK, which is a cooperation between Swedish Environmental Protection Agency (NV), Swedish University of Agricultural Sciences (SLU), and Swedish Meteorological and Hydrological Institute (SMHI).

Dimension	Extent*	Support*	Coverage*
Spatial	Sweden (450 000 km^2)	1000 subbasins (200-700 km^2)	100%
Temporal	Normalised annual average	Daily time-series (15-20 years)	100%

* Terminology according to Bierkens *et al.*, 2000.

itation and evaporation on a part, which represents lakes, rivers and other wet areas. The function consists of one upper, non-linear, and one lower, linear, reservoir. These are the origin of the quick (superficial channels) and slow (base-flow) runoff components of the hydrograph.
Driving model variables are daily precipitation and temperature. These are achieved from optimal interpolation (i.e., kriging) of climate observations considering topography, wind speed and direction in a national grid of 4x4 km (Johansson, 2000; 2002). In the model, subbasins can be disaggregated into elevation zones (for temperature corrections) and land-cover types.
One of the most important parts of the HBV model is the soil moisture routine, which is based on the oversimplified bucket approach, but with the very important additional condition that the water holding capacity of the soil in the subbasin has a statistical distribution (Bergström & Graham, 1998). This leads to a contributing area concept as concerns runoff generation. Only those parts that have reached field capacity will contribute to runoff in the event of rain or snowmelt. It is very important to note that this approach thus implicitly accounts for the subbasin variabilities in both soil water holding properties and input in the form of rain or snowmelt, without explicit separation of the two. The parameter values of the model thus reflect the physical properties of the ground as well as their statistical distribution, and they also reflect the random character of the input. It is similar to the cumulative distribution function used for soil moisture saturation in the ARNO rainfall-runoff model (Todini, 1995), an approach that has also found its way into climate modelling (Dümenil and Todini, 1992) where sub-grid variability is a critical issue.
The application of Sweden includes about 1000 subbasins, ranging in size between 200 and 700 km^2. The model is calibrated regionally against measured time-series of water discharge.

Up-scaling of land cover, emissions and atmospheric deposition

For each subbasin land cover is aggregated into the classes: arable field-type (13 crops on 9 soils in 22 regions; i.e., 2574 types), forest type (3 types), clear-cut forest (additional leaching according to atmospheric deposition rate), urban, and lakes (3 types according to position in the catchment). Emissions are classified as industrial point sources, municipal treatment plants, and rural households. The first two are based on empirical data, while the latter is based on population statistics and coefficients considering average treatment level in the region. The emissions are aggregated into one value for each type and subbasin. Atmospheric deposition is calculated for each lake surface by using seasonal results from the MATCH model (Langner *et al.*, 1995) and aggregated for each lake type (20x20 km; up- or downscaling depending on lake size).

Up-scaling of nitrogen removal processes

The HBV model calculates average storage (and residence-time) of water and N between root-zone and stream, in rivers and in lakes for each subbasin. In the N-routine (Arheimer and Brandt, 1998), leaching concentrations are assigned to the water percolating from the unsaturated zone of the soil to the groundwater reservoir. Different concentrations are used for different land-covers, and the load from rural households is added separately. Removal processes in groundwater are considered before the water and N enter the stream, where additional loads from industry and treatment-plants may be added, as well as river discharge from upstream subbasins. Removal processes may occur during transport in the river and in lakes, and atmospheric deposition is added to lake surfaces (for other land covers it is included in the soil leaching). The equations used to account for

daily removal are conceptual and mainly based on empirical relations between load, temperature and concentration dynamics. The N removal is spatially lumped on a subbasin level into the three categories groundwater, rivers and lakes.

Model calibration and validation

The catchment model includes a number of free parameters, which must be calibrated against time-series of daily observations. The parameter values (coefficients) are tuned to minimise the relative volume error and to maximise the explained variance. About 10 parameters are calibrated for the calculation of water discharge, and 5 to simulate N removal. Calibration is done simultaneously for several observation sites in a region to get robust parameter values, which are then transferred to all subbasins in that region. For N, the calibration procedure is made step-wise, starting with parameters for groundwater, then rivers and finally lakes (Pettersson *et al.*, 2001). Both calibration and validation is done on a daily basis at the subbasin outlet.

In the TRK application covering Sweden, water flow was calibrated against measured daily discharge at the outlet of 230 subbasins, and independent time-series from another 130 subbasins were used for model validation. For N concentrations, time-series from 300 subbasin were used for calibration, while 200 subbasins were used for independent validation. This procedure resulted in a spatial validation of water flow, N-concentrations and transport in the river, according to the proxy-basin concept (Abbott and Refsgaard, 1996).

Monthly grab samples were normally available for N concentrations in rivers, but most time-series only covered part of the period studied. If possible, both water discharge and N concentration were calibrated on a daily basis for the period 1987-1997, and validated for the periods 1983-1986 and 1998-1999. Thus, the temporal dynamics in the model was validated by split-sample test of independent daily time-series.

Up-scaling of results to national level

Source apportionment for different coast segments or for the entire nation is achieved by adding sources for different categories in all subbasins. This is done separately for gross and net loads to illustrate the influence of removal processes. Net load is the remaining part of the gross load, which eventually reaches the sea after the cumulative N removal in groundwater, rivers and lakes downstream a specific source and subbasin (Wittgren and Arheimer, 1996).

Results and Discussion

Model results

The model produces time-series that give the daily variation in water flow, N concentrations and N transport. The time-series show rather good agreement with measured values (Figure 4), both regarding levels and dynamics. In general, it is easier to achieve good correspondence at large river outlets than for individual subbasins. The river flow is regulated by the waterpower industry in most Swedish rivers, which highly influences the dynamics of discharge, especially in the northern part of the country. The diagram at the right upper corner in Figure 4 shows that the model manages to reproduce the general hydrograph, but not the intensive fluctuation in water release for energy production.

The results are spatially distributed as results are achieved from each subbasin included in the modelling. Mapping of the results from the TRK application gives the spatial distribution for the whole country. Figure 5 show the spatial distribution of annual water discharge, as well as the difference between a dry and a wet year. This information

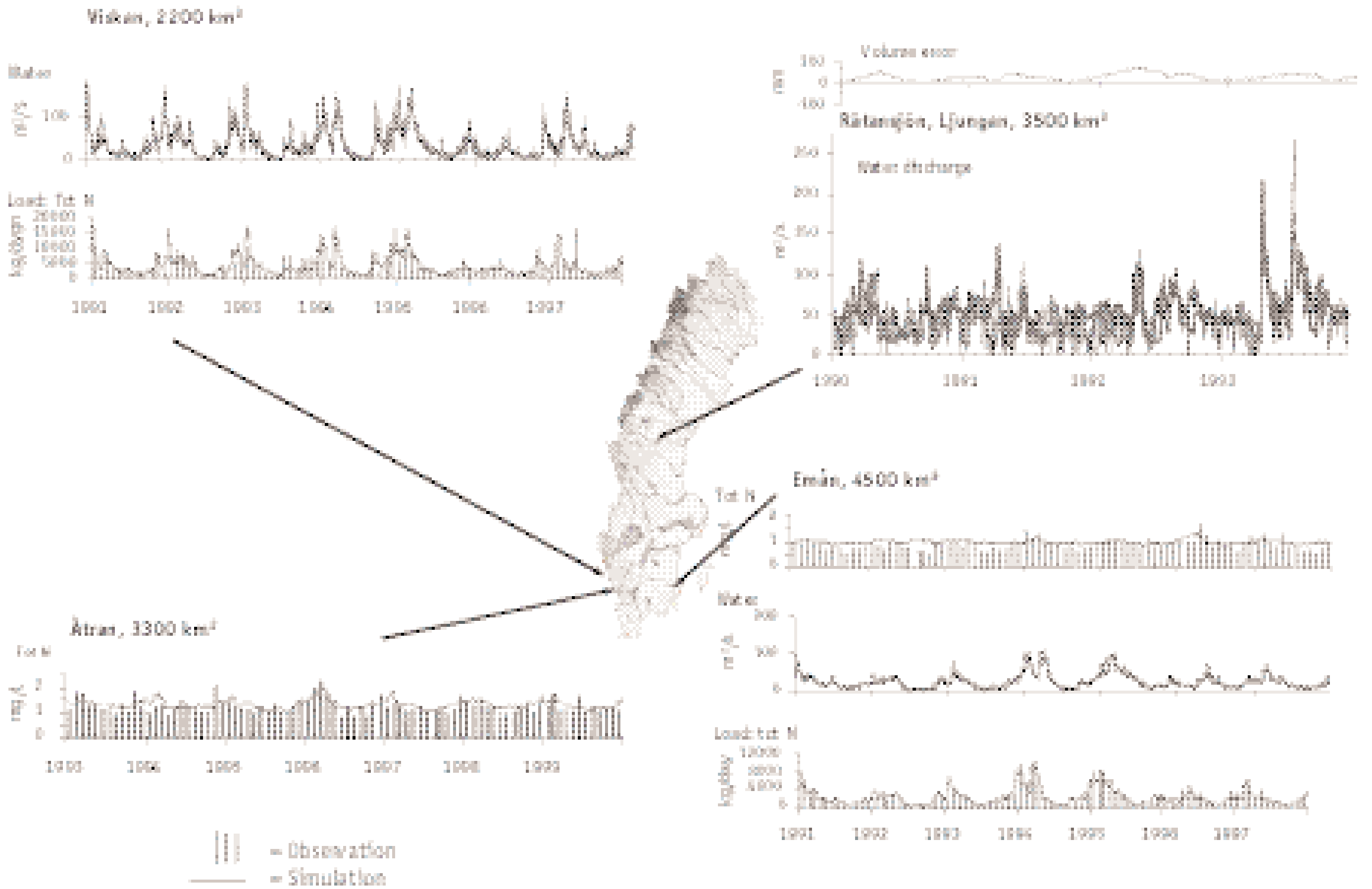


Figure 4. Model performance of simulated time-series compared to observed values (bars). The figure shows examples of independent validation sites, i.e., these time-series were not included in the model calibration procedure.

is important in environmental studies when comparing the nutrient export from one time with another, so that proper flow normalisation is considered to avoid weather impact on the judgement of anthropogenic impact. Similar maps as in Figure 5 will be produced for each year back to 1961, and the modelled time-series are prolonged every year so that the database is up-dated continuously.

The spatial variation in gross N load follows to some extent the pattern of water discharge (cf. Figure 5 and Figure 6) with higher load in the western part of the country. However, the pattern of N soil leaching also reflects the regions in Sweden with most intensive agriculture. For instance, the most southern part of Sweden does not have very high water discharge, but releases the highest N load (Figure 6B). When comparing gross load and net load it can be concluded that in general about 40-50% of the total N load in southern Sweden is removed during transport from the sources towards the sea. However, this downstream reduction in load is not equally distributed but depends very much on the lake distribution of the region and the character of the catchment area and river network downstream the sources. Some areas with intensive agriculture and some major inland point sources do not contribute very much on the N load to the sea (cf. Figure 6B and Figure 6C), while the south-western part has low N retention capacity and still contributes a lot to the total load. When comparing the contribution from various sources (Figure 6A) it can be concluded that the load from arable land is by far the largest source, although the N retention is also high on this load.

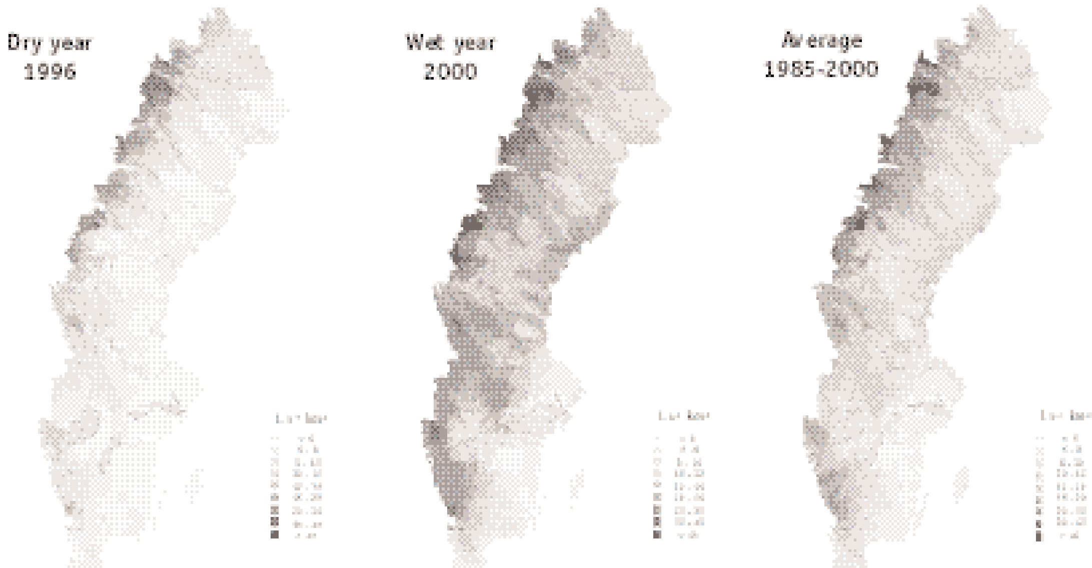


Figure 5. Swedish annual water discharge 1985-2000, according to HBV modelling (modified from Grahn *et al.*, 2002).

Handling scales

Temporal scaling is done when the results are presented as aggregated values. These are based on time-series of 20-30 years with a daily time-step to consider weather-induced variability. An average value for the entire period is considered as normal, i.e., it is assumed not to be affected by specific short-term variations between days, seasons or years. All dynamic modelling of hydrology should be done for at least 20 years if averages are to be considered representative for Swedish conditions. Previous studies show that ten years time-series is not enough to avoid natural hydrometeoroloical variations (Andersson and Arheimer, 2001).

Aggregated values are requested to separate human impact from natural variations. However, during this up-scaling procedure information is lost that may be of critical concern for environmental management. Extreme values of water quality may have severe impact on biology although they appear rarely. Thus, in some situations the extreme situations or seasonal concentrations are of more importance than average conditions. For instance, the daily situation may be of great concern in order to make forecasts on algae concentration close to beaches in the summer time.

Statistical soil moisture distribution and water recharge, along with adding, delaying and subtracting loads along the river course mainly does spatial scaling in HBV-N. The hydrological model accounts for transport and decay within subbasins, and routing through the river system, e.g., when passing lakes, towards the sea. Removal of N may occur during the transport from the sources to the recipient, especially during residence in various water storages, which is considered in the model. The model concept is the same when applied on small river basins and the entire Baltic basin, but the model parameters must be recalibrate when changing the subbasin size. The parameter values of the model reflect the physical properties of the ground, statistical distribution, as well as the random character of the input. The values of the parameters in different basins will therefore be identical as long as the basinwide distribution functions are the same. The model will then be independent of, or at least only mildly sensitive to scale (Bergström and Graham, 1998). This means that to some extent the handling of scales is taken care of within the basic hydrological model concept. Nevertheless, the parameter values consider variability of the environmental conditions and are thus scale dependent.

Once the division into subbasins has been made when setting up the HBV-N model, there is no further spatial resolution and both sources and flow paths are lumped. For analyses on a more detailed scale, new subbasin division

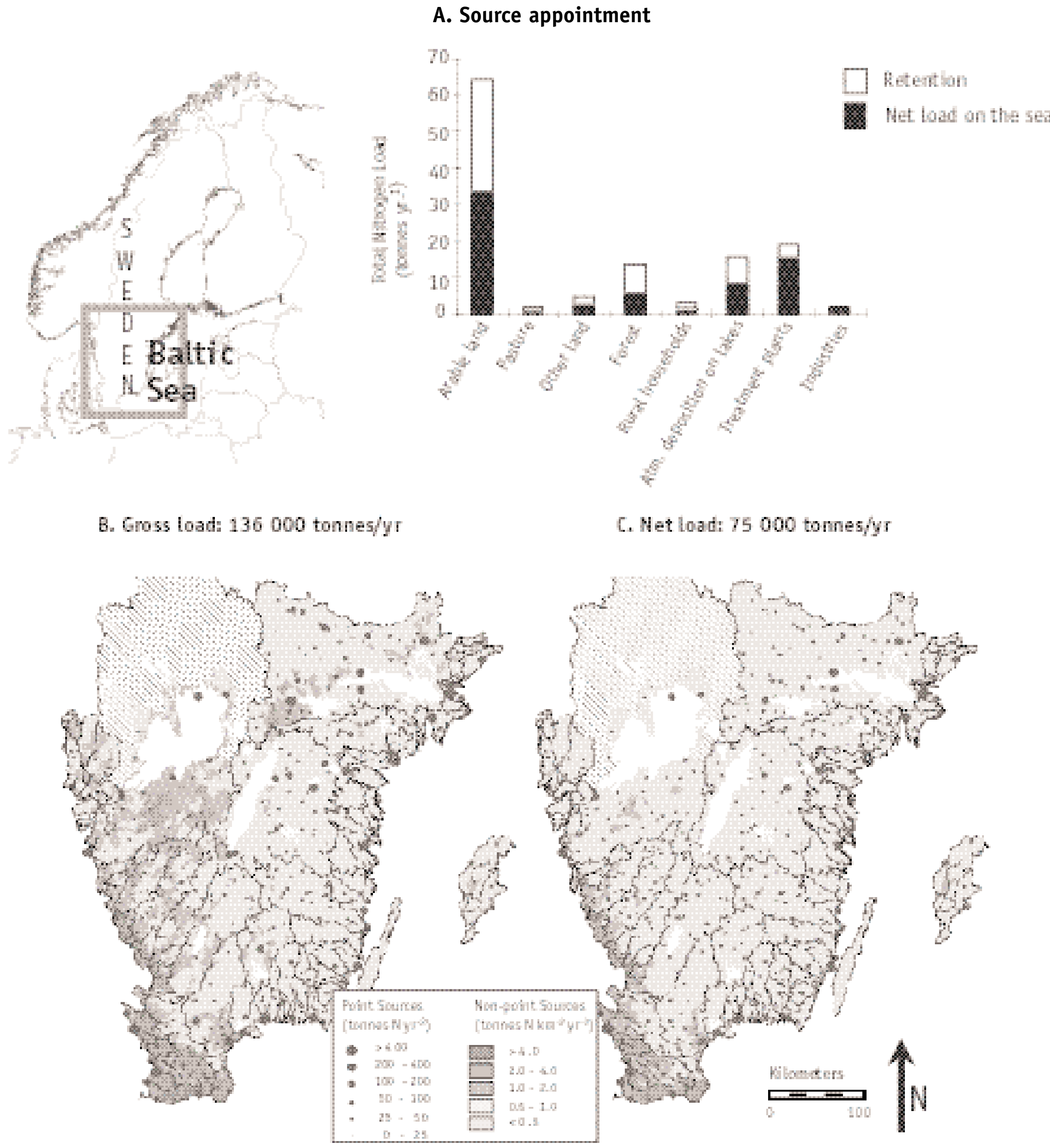


Figure 6. Annual nitrogen transport from land to sea for the southern half of Sweden, based on catchment modelling with HBV-N: A.) the contribution from various sources (i.e., source apportionment); B.) gross load from diffuse and point sources, respectively; C.) net load after nitrogen removal in the fresh-water system between sources and the river outlet (modified from Arheimer and Brandt, 1998)

must be made and the model must be recalibrated against observed values at the new spatial level. The resolution must thus be adapted to the environmental issue in question. As shown in Figure 2, the HBV-N model has been applied at various scales depending on modelling purpose. However, restrictions in site specific information, e.g. precipitation or observation sites for calibration, normally makes very detailed modelling less reliable. It is not advised to apply the HBV-N model for subbasins less than 1 km² if the regular national Swedish databases are used as input data.

Conclusions

• Handling of scales in the HBV-N model is mainly done through up-scaling procedures combined with the basic hydrological model concept. The model is rather insensitive to scale, but parameter values that consider spatial variability of environmental conditions may be scale dependent.

• Temporal and spatial resolution should be adjusted to the purpose with the modelling, as information gets lost at up-scaling. However, it is important that the model can be validated at the highest resolution for which results are presented.

• Integrated catchment models are useful tools in eutrophication management for estimation of nitrogen sources and sinks in the landscape. The coupling of rainfall-runoff models (e.g., HBV) with detailed, field-scale models (e.g., SOIL-N) and GIS may estimate nitrogen load over a range of scales.

Acknowledgements

The following persons contributed to the TRK application; I) From the Swedish Meteorological and Hydrological Institute: Berit Arheimer, Marie Bergstrand, Maja Brandt, Gun Grahn, Anders Gyllander, Barbro Johansson, Lotta Pers, Anna Pettersson, Peter Svensson. II) From the Swedish University of Agricultural Sciences: Heléne Ejhed, Hans-Björn Eriksson, Holger Johnsson, Bert Karlsson, Stefan Löfgren, Kristina Mårtensson, Jakob Nisell, Kjell Olsson, Barbro Ulén. III) From the Swedish Environmental Protection Agency: Anders Widell. IV) From IVL-Swedish Environmental Institute: Olle Westling.

Abstract

There is a request in Sweden of useful tools for more efficient international reporting of nutrient load, and also for eutrophication management and control planning. An integrated catchment model (HBV-N) has therefore been developed. The model has been applied for the national scale (450 000 km²) within a nested model system, called TRK, in which several models with different levels of process descriptions are linked together. Dynamic and detailed models are included for arable leaching, water balance, and N removal. Landscape information, leaching rates and emissions are combined through GIS. The HBV-N model calculates nutrient load, N retention and source contribution to the sea with a relatively high spatial and temporal resolution. The transfer between scales is mainly handled through up-scaling procedures, combined with the basic HBV hydrological model concept. The model is rather scale insensitive, but temporal and spatial resolution should be adjusted to the purpose of the modelling, input data available and possibilities for calibration and validation. The model is validated against monitored time-series of water discharge and nitrogen concentrations. The results show that integrated catchment models are useful tools in eutrophication management for estimating nitrogen sources and sinks in the landscape.

References

Abbott, M. B., and Refsgaard, J. C. (Eds.), 1996. Distributed hydrological modelling, Kluwer Academic Publishers, Amsterdam, 320 pp.

Andersson, L. and Arheimer, B., 2001. Consequences of changed wetness on riverine nitrogen – human impact on retention vs. natural climatic variability. Regional Environmental Change 2:93-105.

Arheimer, B. and Brandt, M., 1998. Modelling nitrogen transport and retention in the catchments of Southern Sweden. Ambio 27(6):471-480.

Brandt, M. and Ejhed, H., 2003. TRK-Transport, Retention, Källfördelning. Belastning på havet. Swedish Environmental Protection Agency, Report No. 5247. (in Swedish)

Bergström, S., 1995. The HBV model. In Singh, V. P. (ed.) Computer Models of Watershed Hydrology, Water Resources Publications, Littleton, Colorado, pp. 443-476.

Bergström, S. and Graham, P., 1998. On the scale problem in hydrological modelling. Journal of Hydrology 211:253-265.

Bergstrand, M., Brandt, M., Arheimer, B., Grahn, G., Gyllander, A., Pers, C., Svensson, P., Ejhed, H., Johnsson, H., Olsson, K., Mårtensson, K., Löfgren, S., and Westling, O., 2002. TRK-nutrient load in Sweden – an operational system for catchment modelling of nutrient transport, retention and source apportionment. In. Killingtveit, Å. (Ed.) Proceedings of Nordic Hydrological Conference, Nordic Hydrological Programme (NHP) Report 47(1): 211-220.

Bierkens, M. F. P., Finke, P. A. and Willigen, P. De. (Eds.), 2000. Upscaling and downscaling methods for environmental research. Kluwer Academic Publishers, London. 190 p.

Cloern, J., 2001. Our evolving conceptual model of the coastal eutrophication problem. Mar. Ecol. Prog. Ser. 29, 280-329.

Dümenil, L. and Todini, E., 1992. A rainfall-runoff scheme for use in the Hamburg climate model. Advances in theoretical hydrology. J. P. o´Kane, (Editor), Elsevier, 129-157.

Grahn, G., Gyllander, A., Johansson, B., and Svensson, P., 2002. Runoff map of Sweden – a method for continous production. In. Killingtveit, Å. (Ed.) Proceedings of Nordic Hydrological Conference, Nordic Hydrological Programme (NHP) Report 47(2): 491-496.

Johansson, B., 2000. Areal precipitation and temperature in the Swedish mountains. An evaluation from an hydrological perspective. Nordic Hydrology, 31:207-228.

Johansson, B., 2002. Estimation of areal precipitation for hydrological modelling in Sweden. Earth Science center, Göteborg University, Report Doctoral thesis No. A76

Johnsson, H., Bergström, L., Jansson, P.-E. and Paustian, K., 1987. Simulated nitrogen dynamics and losses in a layered agricultural soil. Agric. Ecosystems Environ. 18, 333-356.

Johnsson, H., Larsson, M.H., Mårtensson, K. and Hoffmann, M., 2002. SOILNDB: A decision support tool for assessing nitrogen leaching losses from arable land. Environmental Modelling and Software, 17:505-517.

Langner , J., Persson, C. and Robertson, L., 1995. Concentration and deposition of acidifying air pollutants over Sweden: Estimates for 1991 based on MATCH model and observation. Water Air Soil Poll. 85:2021-2026.

Larsson, U., Elmgren, R. and Wulff, F., 1985. Eutrophication and the Baltic Sea – Causes and consequences. Ambio 14:9-14.

Lindström, G., Johansson, B., Persson, M., Gardelin, M., and Bergström, S., 1997. Development and test of the distributed HBV-96 hydrological model, J. Hydrol., Vol. 201, pp. 272-288.

Lääne, A., Pitkänen, H., Arheimer, B., Behrendt, H., Jarosinski, W., Lucane, S., Pachel, K., Räike, A., Shekhovtsov, A., Svendsen, L.M., and Valatka, S., 2002. Evaluation of the implementation of the 1988 Ministerial Declaration regarding nutrient load reductions in the Baltic Sea catchment area. The Finnish Environment Institute (FEI), The Finnish Environment Report No. 524. Helsinki. pp 195.

Pettersson, A., Arheimer, B. and Johansson, B., 2001. Nitrogen concentrations simulated with HBV-N: new response function and calibration strategy. Nordic Hydrology 32(3):227-248.

Stålnacke, P., Vandsemb, S.M., Vassiljev, A., Grimvall, A. and Jolankai, G., 2002. Changes in nutrient levels in some Eastern European rivers in response to large-scale changes in agriculture. Proceedings of 6th International Conference on Diffuse Pollution, International Water Association (IWA), Amsterdam, pp 267-274.

Swedish EPA, 1997. Nitrogen from land to sea. Main report. Swedish Environmental Protection Agency, Report 4801, Nordstedts tryckeri AB, Stockholm.

Todini, E., 1995. The ARNO rainfall-runoff model. J. Hydrol., 175:339-382.

Wittgren, H. B. and Arheimer, B., 1996. Source apportionment of riverine nitrogen transport based on catchment modelling. Water Science and Technology 33(4-5):109-115.

Nutrient fluxes at the river basin scale

MARCEL DE WIT

Dr. M.J.M. de Wit, RIZA, P.O Box 9072, 6800 ED Arnhem, The Netherlands.
m.dwit@riza.rws.minvenw.nl

The impact of nutrient pollution can be observed in many rivers and coastal seas all over Europe (Stanners & Bourdeau, 1995). This has led to international directives that aim at a reduction of nutrient levels in European rivers and coastal seas (EEC, 2000). Large scale studies of nutrient fluxes are needed to predict and evaluate the effects of the proposed measures.

For most applied environmental research the scale (extent and resolution) of the available data (information scale) differs from the scale at which most of the underlying processes typically occur (model scale) and the scale at which the outcome of the research is used (policy scale). Therefore, upscaling and downscaling methods (Bierkens *et al.*, 2000) are often an essential part of environmental research (e.g. Feddes, 1995; Addiscott, 1998). However, the use of scale transfer functions does often not improve the transparency of the linkage between question (policy scale) and answer (policy scale). The aim of this paper is to illustrate that before using scale transfer functions to transpose available models and data into the policy scale one may search for data and model concepts that match the policy scale. This is demonstrated with examples derived from the analysis of Nitrogen (N) and Phosphorus (P) fluxes in the Rhine and Elbe river basins (Figure 1). The search for an appropriate model to analyse nutrient fluxes at the river basin scale involves consideration of the spatial and temporal extent and resolution needed to ans-

Figure 1 The Rhine and Elbe river basins cover an area of approximately 300,000 km2 of which about 45 percent is used for agricultural production. The two river basins have a total population of around 70 million people and they overlap with the borders of 11 different countries. Together these basins cover a wide range of landscape, climatic, and socio-economic zones. The Nitrogen and Phosphorus fluxes in these rivers have increased with time by human activities. This has caused considerable changes in fresh and marine ecosystems and has negatively affected the quality of water for human consumption and other uses (Stanners & Bourdeau, 1995)

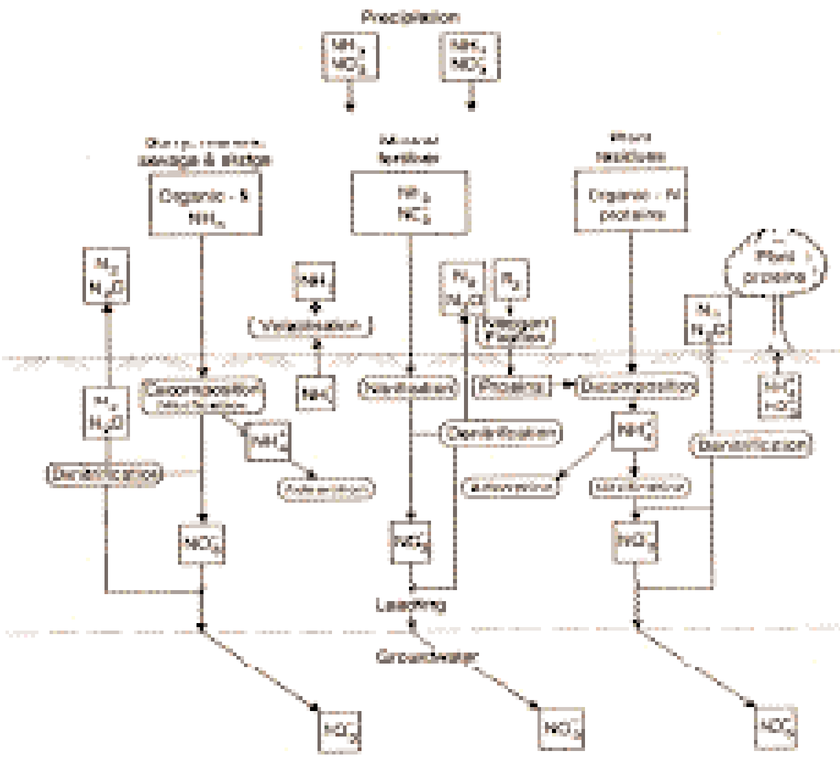


Figure 2. Processes that determine the flux of nitrogen in the soil (Burt *et al.*, 1993)

wer the questions that are relevant for nutrient policy at the river basin scale, and the availability of data to cover the extent of the study at the required resolution. Furthermore, it needs to be determined which factors are the main controls of nutrient fluxes at the river basin scale and how these factors should be represented in the model given the quality and resolution of the available data.

A matter of scale

In applied environmental studies research questions are often scale specific. The scale of the research largely determines which method is appropriate to use. Therefore, it is necessary to explicitly define the scale of the research before choosing the methodology. One should consider both the spatial and temporal extent and the spatial and temporal resolution needed to answer the question. In general the larger the extent, the less detailed is the resolution, that is considered for the analysis. At a regional scale nutrient fluxes are not analysed to learn about microscopic processes in the soil, but rather to describe long term and regional patterns. Also, the resolution of the available data generally decreases with increasing size of the study area. For the analysis of nitrogen leaching at the scale of a farm one might use data from field experiments, whereas for a regional analysis of nitrogen pollution one has to work with soil maps and regional administrative data. Finally, different factors dominate at different levels of scale (see for example Figure 2). Temperature might be one of the main variables to describe the variation in N concentrations within a year, but it is of much less importance for the description of the variation between different years.

The framework presented in Table 1 summarises the foregoing discussion and was used to develop a modelling strategy for the analysis of nutrient fluxes from pollution sources to the river outlets at the river basin scale. The nutrient study described in this paper aims at answering two questions: 1) what is the contribution of the different sources (agriculture, industry etc.) and regions to the nu-

Table 1. Examples of the analysis of nutrient fluxes at different levels of scale

Research aim	Extent of research	Resolution	Available data	Dominating factors
understanding processes	point	detailed	laboratory experiments	denitrification,adsorption
protecting the trophic status of a small lake	small region: decade	hectare: month	stream flow data, field measurements	agricultural practices, flow velocity
global/climate change	world: century	country: year	administrative data, climate data	population density, economy

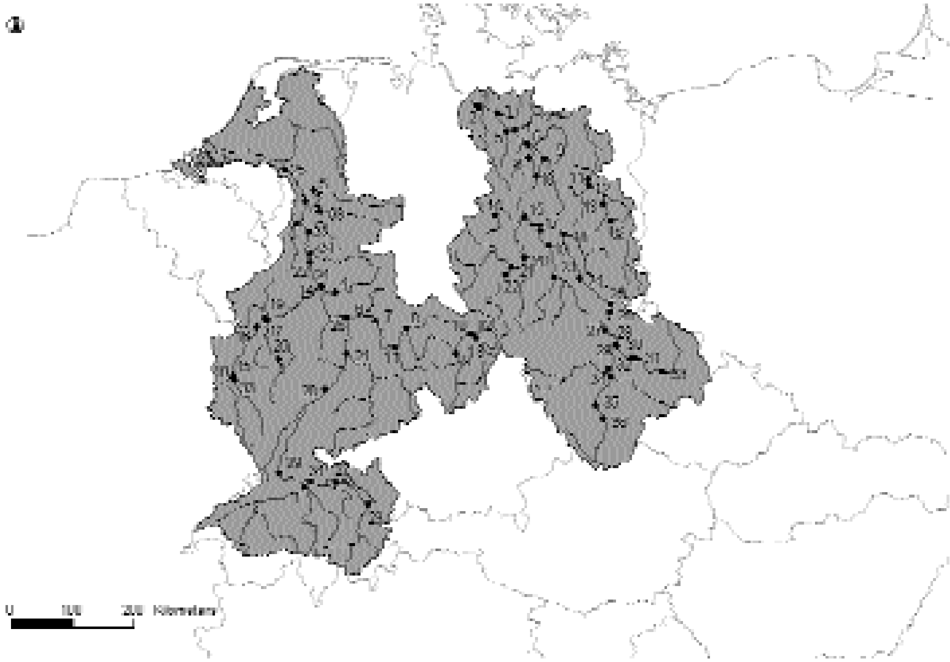


Figure 3 Location of monitoring stations used in this study

trient load in the river?, and II) what will be the effect of source control measures (e.g. reduction of fertiliser use or the improvement of wastewater treatment plants) on the nutrient load in the river? For large areas such as the river basins analysed in this study (10^5 km²) these questions need to be evaluated over long time periods (decades rather than years). From a (European) policy point of view a spatial resolution of 10^3-10^4 km² (upstream basins of major tributaries) and a temporal resolution of five years are a reasonable resolution to analyse the past (since 1970) and future (up to 2020) changes in nutrient sources and nutrient loads in the Rhine and Elbe river networks.

The next step is to explore what data are available at the scale (extent and resolution) of the research question. It appeared that for the analysis of nutrient fluxes in the Rhine and Elbe basins a lot of data were available that cover the entire river basins and have the required (or even more detailed) resolution. An overview of the data available for the analysis of nutrient emissions and nutrient transport (from pollution sources to river outlets) are given in Table 2. Water quality and water quantity data were available for 70 stations spread over the Rhine and Elbe river networks (see Figure 3). The area upstream of these monitoring stations varies between 10^3-10^5 km². These data were available to calibrate and validate the models. More detail about the data used for the nutrient study is given in De Wit (1999a).

The quality and resolution of the available data should be seen as a precondition for the type of model to be developed and not as an excuse afterwards for why an advanced and intricate model does not perform well. There is no point in using a model for which the appropriate data are not available. Moreover, the model should consider those factors that dominate at the scale of the analysis. The question now is: which factors are dominating for nutrient fluxes at the river basin scale and how should these factors be represented in the model given the quality and resolution of the available data?

The balance between data availability and model complexity

The search for an appropriate model at the river basin scale was done by comparing the results of four different models that represent increasing complexity (De Wit & Pebesma, 2001). In the first model only one variable is used, in the second model two variables are used, in the third model three variables are used and in the fourth model a large number of variables are included. The five year average N and P loads measured at 34 different mon-

itoring stations in the river Rhine network (1970-1995) were used to calibrate the models. The model parameters were tuned in such a way (trial and error) that the difference between measured and modelled five year average river load was minimised. The five year average N and P loads measured at 36 different monitoring stations in the river Elbe network (1980-1995) were used to validate the models. A comparison of the predictive capability of the four models can be used to determine the utility of increasing model complexity. The errors in the data that were used to run and validate the models were quantified and it was analysed to what extent the model validation errors could be attributed to data errors, and to what extent to shortcomings of the model. For more details the reader is referred to De Wit & Pebesma (2001).

Table 2. Data available for the analysis of nutrient fluxes at the river basin scale

Data available for the analysis of nutrient emissions

Data	Resolution	Period	Source
Population numbers	Regions	1990-1995	National Statistical Agencies
Connection rate sewage systems	Regions	1990-1995	,,
Connection rate WWTP [a]	Regions	1990-1995	,,
Information WWTP [a]	WWTP [a]	1990-1995	,,
Industrial emissions	Regions	1990-1995	,,
Livestock numbers	Regions	1970-1995	,,
Agricultural land use	Regions	1990-1995	,,
Crop yields	Regions	1990-1995	EUROSTAT [b]
Crop yields	Country	1970-1995	FAO
Fertiliser use	Country	1970-1995	FAO
Land Cover	1 km^2	1990-1995	Corine, USGS [c]

Data available for the analysis of nutrient transport in soil, groundwater, and river network

Data	Resolution	Period	Source
Average annual precipitation	9 km^2	long term	PIK [d]
Average annual temperature	9 km^2	long term	PIK [d]
Soil type	1:1 M	-	ESB [e]
Lithology	1:1 M	-	Derived from soil map, IAH [f]
Elevation	1 km^2	-	USGS [c]
Slope (relief)	1 km^2	-	Derived from elevation map
River network (LDD[g])	1 km^2	-	Derived from elevation map

[a] Wastewater treatment plant
[b] European Statistical Office
[c] United States Geological Survey
[d] Potsdam Institute for Climate Impact Research
[e] European Soil Bureau
[f] International Association of Hydrogeologists
[g] Local drain direction map

In the first model it is assumed that the five year average river load at a certain monitoring station in the river network and for a certain time period (e.g. 1970-1975 or 1990-1995) is proportional to the size of the upstream basin. This model serves as a starting point. It lacks any description of the upstream basin. It represents the level of knowledge that was available before the analysis of pollution sources and transport conditions in the Rhine and Elbe basins.

The second model is based on the assumption that the river load is proportional to nutrient emissions in the upstream basin or in other words: 'the larger the nutrient input the larger the nutrient output'. A distinction is made between direct nutrient emissions to the surface water (e.g. discharge of wastewater) and nutrient surplus at the soil surface (input from fertilisers, manure, and atmospheric deposition minus output from yield). Both were mapped for the entire Rhine and Elbe basins at a resolution of 1 km^2 for all five year periods from 1970 to1995 (see De Wit, 1999a) and have been used as input for models two, three and four. The ratio of transport of nutrients through the soil/groundwater system and the ratio of transport through the river network are constant in this model for all regions and time periods. This second model represents the level of knowledge available after the inventory of pollution sources and before the analysis of transport conditions.

The third model (De Wit, 1999b) describes the ratio of transport of nutrients through the river network as a function of the area specific runoff. The ratio of transport of

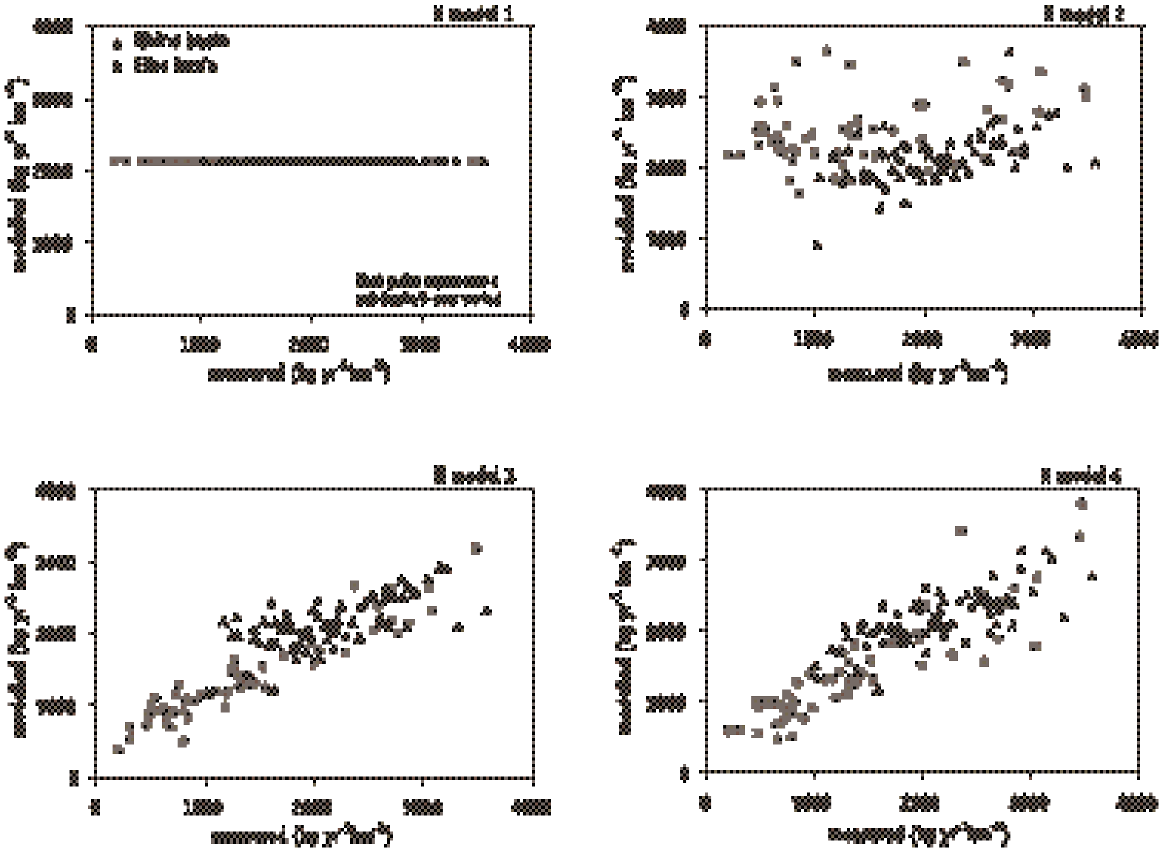


Figure 4 Measured and modelled area specific nitrogen load. The figure shows that the model performance increases when moving from model 1 to model 3. The shift to model 4 does not improve the model outcome. Similar results were obtained for phosphorus. For more details about the performance of the four models, the reader is referred to De Wit & Pebesma (2001).

nutrients through the soil/groundwater system is described as a function of lithology. Here, a different parameter value is used for regions with consolidated and regions with unconsolidated rocks. This model describes the river nutrient load as a function of nutrient emissions in the upstream basin, where the fraction of the nutrients that reaches the outlet of the river is positively related to runoff, and the ratio of transport through the soil/groundwater system is larger for regions with consolidated rocks than for regions with unconsolidated rocks.

The fourth model is a conceptual model that is described in detail in De Wit (2001). It is linked to a GIS environment. The fraction of the nutrient surplus at the soil surface that leaches, erodes, volatises or is stored in the soil/groundwater system is related to the total runoff, groundwater recharge, groundwater travel times (see De Wit *et al.*, 2000), slope, soil type, and aquifer type at each specific location (km^2). Dynamic functions are used to account for the delay of nutrient transport in the soil and the groundwater. A drain direction map is used to route the nutrients through the river network. In each river segment (1 km) a certain fraction of the nutrient load is lost, depending on the flow regime in the specific cell.

From a comparison of the models, it was concluded (De Wit & Pebesma, 2001) that although the addition of more process description is interesting from a theoretical point of view, it does not necessarily improve the predictive capability. Although the analysis is based on an extensive pollution sources-river load database (see Table 2) it appeared that the information content of this database was only sufficient to support a model of a limited complexity. However, this model (model three) successfully described most of the observed spatial and temporal variation in nutrient fluxes at the river basin scale. Moving from model one to model two to model three appeared to be improvements. The step from model three to model four did not yield better simulations of nutrient fluxes (see Figure 4).

The balance between the quality of the available data and the complexity of the model had been reached.

Discussion

A challenging aspect of this study is its spatial and temporal extent; river basins of the order 10^5 km^2 and a time period of interest of 50 years. The pathways, and fate of nutrients in soil, groundwater, and river network are a complex function of biological, chemical, and physical processes. Nonetheless it appeared to be possible to simulate most of the observed spatial and temporal variation of nutrient fluxes in the Rhine and Elbe basins. This good result can be attributed to the following points:

- A consideration of the resolution needed to answer the research question resulted in the choice to model at a temporal resolution of five-year periods. This resolution is detailed enough to monitor the effects of large scale policy and very much simplified the analysis since short-term variation in nutrient fluxes were not considered. In the same way the use of a less detailed spatial resolution may prevent the researcher from being drawn in small scale variation that are not relevant at the scale of an entire river basin.
- The search for data at the river basin scale was more successful than expected. Due to advances in technique there is a growing amount of digital spatial data available for environmental research. Data derived from satellite images, supranational mapping programs (e.g. Corine, European soil map), uniform administrative data (e.g. Eurostat), and long term monitoring programs (e.g. water quality monitoring) continuously offer new opportunities for the modelling of environmental issues (Burrough & Masser, 1998).
- The relatively good performance of model three in the analysis of nutrient fluxes shows that most of the spatial and temporal variation in nutrient loads in the river Rhine and Elbe can be explained by an inventory of nutrient emissions and a description of the transport of nutrients as a function of two variables; precipitation surplus and lithology. Apparently these two variables are large scale 'surrogate' variables that reflect the most important processes that determine the pathways and fate of nutrients from pollution sources to river outlets.

An alternative to the method presented in this paper would have been to use existing process-based models for water and nutrient fluxes in soil, groundwater, and rivers and combine these models (using scale transfer functions) to derive a tool that can be used for the entire river basin. It would be interesting to compare the results of such a methodology with the results of the river basin models (three and four) presented in this paper. Such a comparison is however, beyond the scope of this study.
The message of this paper is that before using scale transfer functions to transpose available models and data into the policy scale one may search for data and models that match the policy scale. This appeared to be a successful approach for the analysis of long-term nutrient fluxes at

the river basin scale and may also be a useful strategy for some other environmental studies. Still, for many other studies (e.g. the influence of global warming on regional flooding) the need for scale transfer functions will probably appear to be unavoidable.

Acknowledgements

Most of the work presented in this paper has been performed at the department of Physical Geography of Utrecht University with support of RIVM. Three anonymous reviewers are acknowledged for their valuable comments.

Abstract

The impact of nutrient pollution can be observed in rivers and coastal seas all over Europe. Much is known about the biological, chemical, and physical processes that determine the pathways and fate of nutrients in soil, groundwater, and surface water. However, there is a large gap between the scale at which these processes typically occur and the understanding of nutrient fluxes at the scale of entire river basins. This paper shows how the scale issue was considered for the analysis of long-term nutrient fluxes in the Rhine and Elbe river basins. Although this analysis is based on an extensive pollution sources-river load database it appeared that the information content of this database was only sufficient to support a model of a limited complexity. Nevertheless, this model successfully described most of the observed spatial and temporal variation in nutrient fluxes in the Rhine and Elbe river basins.

References

Addiscot, T.M., 1998. Modelling concepts and their relation to the scale of the problem. Nutrient cycling in Agroecosystems 50: 239-245.

Bierkens, M.F.P., P.A. Finke & P. de Willigen, 2000. Upscaling and downscaling methods for environmental research. Developments in Plant and Soil Sciences, volume 88. Kluwer, Dordrecht.

Burrough, P.A. & I.Masser, 1998. European Geographic Information Infrastructures: opportunities and pitfalls. Taylor & Francis, London.

Burt, T.P., A.L. Heathwaite & S.T. Trudgill, 1993. Nitrate: processes, patterns and management. Wiley, Chichester.

De Wit, M.J.M., 1999a. Nutrient fluxes in the Rhine and Elbe basins. Netherlands Geographical Studies (NGS): 259. PhD Thesis Utrecht University.

De Wit, M.J.M., 1999b. Modelling nutrient fluxes from source to river load: a macroscopic analysis applied to the Rhine and Elbe basins. Hydrobiologia 410: 123-130.

De Wit, M.J.M., C. Meinardi, F. Wendland & R. Kunkel, 2000. Modelling water fluxes for the analysis of diffuse pollution at the river basin scale. Hydrological Processes 14: 1707-1723.

De Wit, M.J.M., 2001. Nutrient fluxes at the river basin scale. Part I: The PolFlow model. Hydrological Processes 15: 743-759

De Wit, M.J.M. & E.J. Pebesma, 2001. Nutrient fluxes at the river basin scale. Part II: The balance between data availability and model complexity. Hydrological Processes 15: 761-775

EEC, 2000. Council Directive 2000/86/EEC of 23 October 2000 establishing a framework for Community action in the field of water policy. European Commission, Bruxelles.

Feddes R.A., 1995. Space and time variability and interdependencies in hydrological processes. International Hydrological Series. University Press, Cambridge.

Stanners, D. & P. Bourdeau, 1995. Europe's environment. The Dobris Assessment. European Environmental Agency. EEA, Copenhagen

Foto: Joep Dirkx

Assessing the ecological condition of wetlands at the catchment scale

Hydrogeomorphic
Upscaling
Wetland assessment
Catchment
Chesapeake bay

Rapid assessment methods for evaluating the functioning and biodiversity status of wetlands are mostly carried out at the scale of individual wetlands. There is an increasing need for evaluating the condition of wetlands at the watershed scale. We used statistical procedures to determine the relationships between data compiled in field-based assessments of individual wetlands and spatial data from remote sensing or other mapping efforts. The goal was to determine if available geographic data could be used to assess individual wetlands or the overall condition of wetlands in the watershed without having to do site-specific assessments based on field sampling.

DENNIS WHIGHAM, DONALD WELLER, AMY DELLER JACOBS, THOMAS JORDAN & MARY KENTULA

Prof. dr. D. F. Whigham, Smithsonian Environmental Research Center, Box 28, Edgewater, MD 21037, USA. **Dr. D. E. Weller**, Smithsonian Environmental Research Center, Box 28, Edgewater, MD 21037, USA. **A. Deller Jacobs**, The Nature Conservancy of Delaware, 100 West 10th Street, Suite 1107, Wilmington, DE 19801, USA ; Present Address: Delaware Department of Natural Resources and Environmental Control, Division of Water Resources, 820 Silver Lake Blvd, Suite 220, Dover, DE 19904, USA. **Dr. T. E. Jordan**, Smithsonian Environmental Research Center, Box 28, Edgewater, MD 21037, USA. **Dr. M. E. Kentula**, U.S. Environmental Protection Agency, National Health and Environmental Effects Laboratory—Western Ecology Division, 200 SW 35th Street Corvallis, OR 97330, USA.

The movement of water through landscapes is most effectively managed at the level of individual catchments (hereafter referred to as watersheds), and wetlands are important components of watersheds because of their ability to retain, store, and transform nutrients, toxics, water, and sediments that originate from both diffuse and point sources (Whigham *et al.*, 1988; Johnston *et al.*, 1990; Dorioz & Ferhi, 1994; Weller *et al.*, 1996; Greiner & Hershner, 1998; Kuusemets & Mander, 1999; Crumpton, 2001; Reed & Carpenter, 2002). Effective watershed management thus requires knowledge about the abundance, location, and ecological condition of wetlands within the watershed.

Most assessments of wetland condition occur at the level of individual wetlands (Bartoldus, 1999), and few approaches are available to assess the condition of wetlands at the scale of an entire watershed. Wetlands have been considered as elements of watersheds for purposes of risk assessment (Lemly, 1997; Detenbeck *et al.*, 2000; Cormier *et al.*, 2000; Leibowitz *et al.*, 2000.), but this approach does not result in any characterization of wetland ecological condition. Geographic analysis of digital maps has been used to determine the importance of wetlands in reducing nutrient runoff from watersheds (e.g., Weller *et al.*, 1996) and to identify the location of significant wetlands in watersheds (Cedfeld *et al.*, 2000; Crumpton, 2001). While Weller and colleagues were successful in demonstrating the importance of riparian wetlands in reducing phosphorus in surface water, Cedfeld and colleagues had limited success in identifying potentially important wetlands in a watershed because of difficulties in correlating results of the geographic analysis with results from field-based assessments.

If wetland management and restoration are to be successful at the watershed scale, we need analytical methods to evaluate wetland condition, identify important wetlands in watersheds, and determine where wetland restoration efforts should be concentrated (O'Neill *et al.*, 1997). In this paper, we describe an approach that we used to evaluate the ecological condition of two types of wetlands individually and at the scale of an entire watershed. We describe two of the primary goals of the study. The first is to evaluate the condition of wetlands within the watershed by using a field-based assessment approach in combination with a probability-based method for selecting a spatially representative sample. The second goal is to determine if geographic analysis of mapped data can be used separately or in combination with the field-based assessment approach to characterize the condition of individual wetlands or the populations of wetlands in a watershed. In this paper we focus on issues related to selection of assessment sites, the range of assessment scores for both wetland classes at the scale of the entire watershed, and the suitability of using geographic data to conduct site assessments.

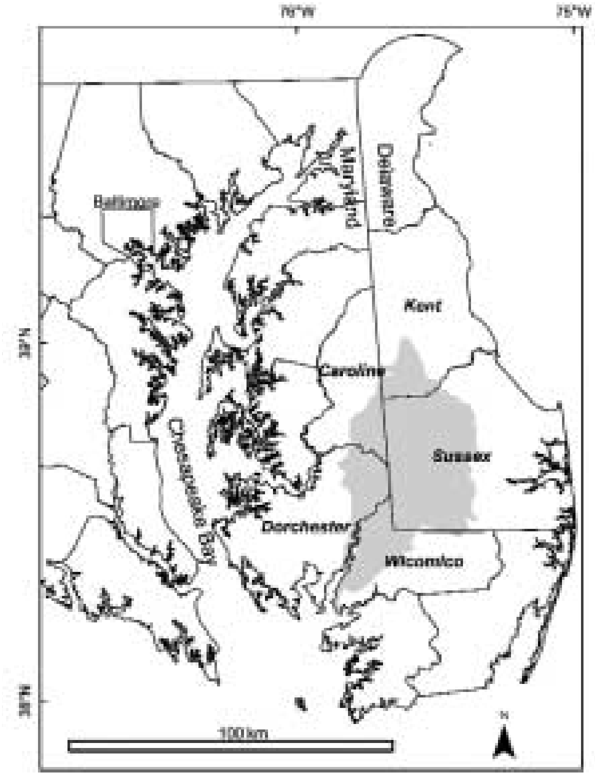


Figure 1. Map of the Chesapeake Bay region showing location of Nanticoke River watershed (shaded area).

Nanticoke River watershed and its wetlands

The Nanticoke River drains approximately 283,000 ha of three counties in Maryland and two counties in Delaware (Figure 1). Agriculture occurs on more than 40% of the watershed and less than 2% has been characterized as urban and suburban development (The Nature Conservancy, 1994). Forests cover approximately 45% of the watershed but many are intensively managed and harvested (Bohlen & Friday, 1997). Agriculture and forest management have been supported by extensive drainage and most nontidal wetland losses in the watershed have been the result of drainage by channelization (Tiner, 1985). Water quality problems are common within the watershed and are mostly related to surface and subsurface runoff from intensive agriculture (e.g., Phillips *et al.*, 1993; Jordan *et al.*, 1997). About 27% of the watershed contains both tidal and non-tidal wetlands (Tiner, 1985; The Nature Conservancy, 1994; Tiner & Burke, 1995). Non-tidal wetlands, the focus of this project, account for almost 85% of all wetland area and are mostly associated with streams (riverine wetlands), poorly drained depressions (depressional wetlands), and poorly drained sites that are relatively flat (flats wetlands).

The Nanticoke watershed is of interest to conservation organizations such as The Nature Convervancy because of the presence of almost 200 plant species and 70 animal species that have been listed as rare, threatened or endangered by the states of Maryland and Delaware (The Nature Conservancy, 1994).

Project Design

The project design integrated three components (Figure 2). First, the hydrogeomorphic (HGM) method for wetland assessment was used to assess the ecological conditions of individual wetlands. Second, the selection of sites for conducting HGM assessments was accomplished by applying methods developed by the U.S. Environmental Protection Agency Environmental Monitoring and Assessment Program (EMAP). Third, GIS procedures were used for two purposes. Selected spatial data were used to assist in the HGM assessments of individual wetlands and a separate effort focused on the potential use of spatial data to assess wetland condition from mapped information.

The hydrogeomorphic (HGM) method (Brinson *et al.*, 1995; Smith *et al.*, 1995; Brinson & Rheinhardt, 1996; Whigham *et al.*, 1999) is one of more than 40 approaches that have been developed in the U.S. to assess wetland conditions

(Bartoldus, 1999). In brief, the method produces Functional Capacity Index (FCI) scores for specific wetland functions. FCI scores range from 0.0 to 1.0 and they are calculated from equations that combine scores for individual variables. Individual variable scores also range from 0.0 to 1.0 and they are quantified by evaluating data collected at the assessment site. Variable scores are determined based on reference sites; the higher the score the more similar a variable is to a site with minimal disturbance. Once models are developed, the HGM procedure is intended to be a fairly rapid assessment, requiring 0.5 to 1.0 day of data collection. Details of the HGM procedures can be found in the references cited above and a list of HGM publications found on a web site maintained by the U.S. Army Corps of Engineers (*http://www.wes.army.mil/el/wetlands/wlpubs.html*).

HGM models specific to the Nanticoke watershed were developed in two phases. The *Developmental Phase* took approximately one year to complete. First, an interdisciplinary team of biologists, soil scientists, and wetland ecologists identified the dominant wetland classes and selected potential variables (Table 1) for use in the HGM models (Table 2). The selection of variables was based on existing knowledge about wetlands in the study area and information available from efforts to develop HGM models for similar classes of wetlands (e.g., Brinson *et al.*, 1995; Whigham *et al.*, 1999; Rheinhardt *et al.*, 2002). The interdisciplinary team then selected a series of Reference Wetlands (Figure 3) to represent the full range of altered and unaltered conditions. These wetlands were sampled using protocols based on the experiences of the interdisciplinary team and procedures published by other groups who had developed HGM models. For riverine wetlands, sampling procedures relied on methods developed by Whigham and colleagues (Whigham *et al.*, 1999) for riverine wetlands in the same region. For flat wetlands, sam-

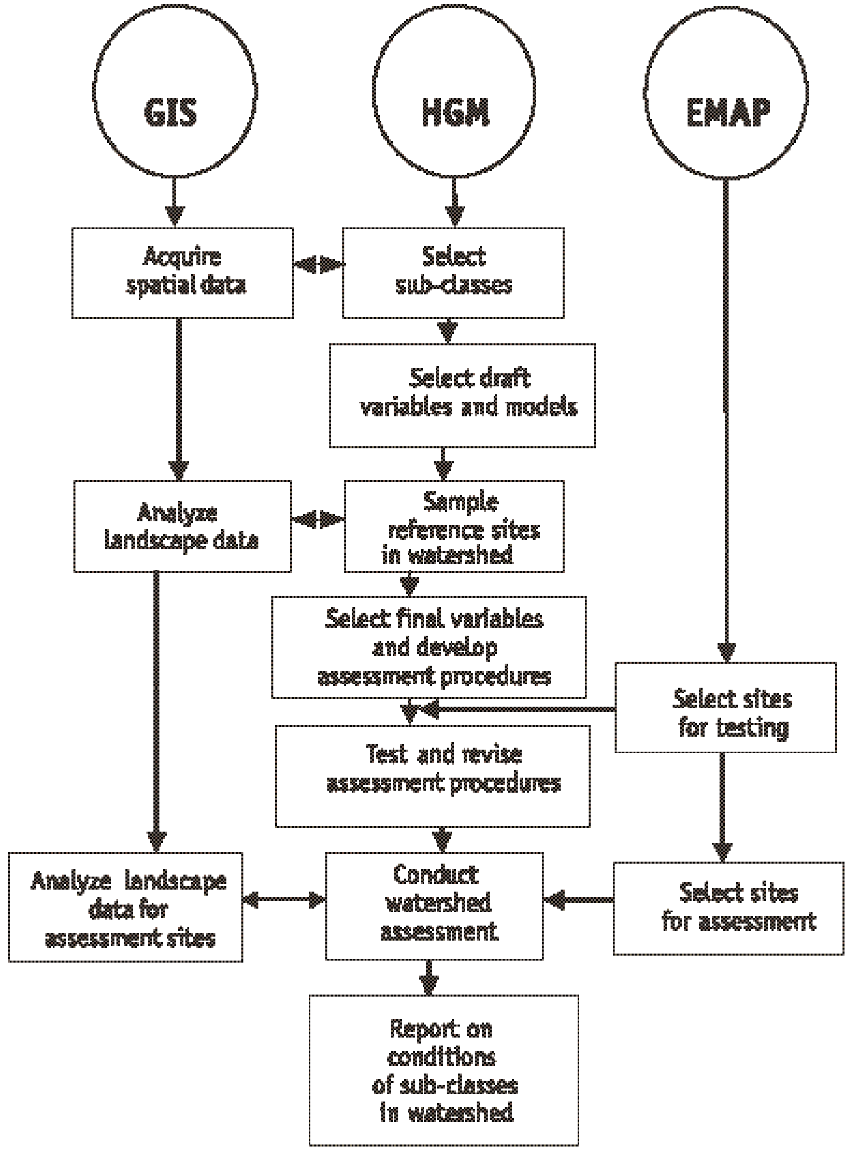


Figure 2. Box and arrow diagram showing the organizational structure of the project. The three elements of the project described in this paper included the development and application of field-based hydrogeomorphic (HGM) assessments, the use of mapped geographic data (GIS), and the sample design provided by the Environmental Monitoring and Assessment Program (EMAP).

Flats Class		Riverine Class	
V_{ANIMAL}	Number of vertebrate species	V_{CANOPY}	Percent tree canopy cover
V_{CANOPY}	Percent tree canopy cover	V_{CWD}	Density of coarse woody debris
$V_{DISTURB}$	**Evidence of vegetation disturbance**	V_{DITCH}	Presence of ditches on floodplain
V_{DRAIN}	**Percent of assessment area affected by drainage**	**$V_{FARBUFFER}$**	**Condition of buffer within 20-100 m**
V_{FILL}	**Presence of anthropogenic derived sediment**	**$V_{FLOODPLAIN}$**	**Floodplain condition**
V_{HERB}	**Species of herbs present**	**V_{HERB}**	**Species of herbs present**
V_{ANTHRO}	Number of anthropogenic features	**$V_{INVASIVE}$**	**Presence of invasive species**
$V_{LANDUSE}$	Land-use of adjacent upland habitats	$V_{LANDUSE}$	Land-use within 1 km of wetland
V_{LITTER}	Percent litter cover	V_{MICRO}	Presence of microtopographic features
$V_{LITTDEPTH}$	Litter depth	**$V_{NEARBUFFER}$**	**Condition of vegetation buffer within 0-20 m**
V_{LOG}	Density of downed logs	V_{ROOT}	Root abundance
V_{MICRO}	**Presence of microtopographic features**	**$V_{SAPLING}$**	**Sapling species composition**
V_{RUBUS}	**Presence of *Rubus* sp.**	$V_{SEEDLING}$	Seedling density
$V_{SAPLING}$	Sapling density	**V_{SHRUB}**	**Shrub density**
V_{SHRUB}	**Shrub density**	V_{STRATA}	Number of vegetation strata
V_{SNAG}	**Density of standing of standing dead trees**	**$V_{STREAMIN}$**	**Stream condition inside assessment area**
V_{STRATA}	Number of vegetation strata	**$V_{STREAMOUT}$**	**Stream condition outside assessment area**
V_{TREE}	**Tree species composition**	**V_{TBA}**	**Basal area of trees**
V_{TBA}	**Basal area of trees**	**V_{TDEN}**	**Tree density**
V_{TDEN}	**Tree density**	**V_{TREE}**	**Tree species composition**
$V_{TREESEED}$	Number of tree seedling species	$V_{TREESEED}$	Number of tree seedling species
V_{VINE}	Number of vine species	**V_{VINE}**	**Number of vine species**

Table 1. Variables considered for inclusion in HGM models for riverine and flats classes in the Nanticoke River watershed. Variables that were chosen for use in the models shown in Table 2 are shown in bold.

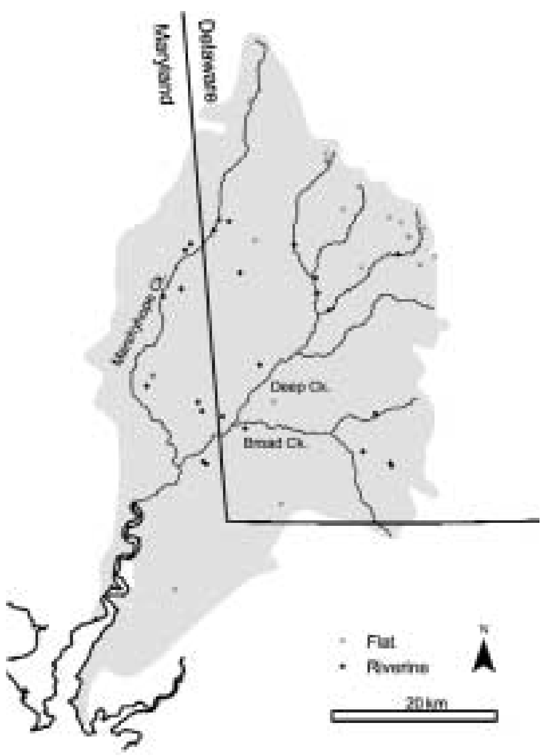


Figure 3. Location of Reference Wetlands within Nanticoke River watershed for riverine and flats subclasses.

pling procedures were based mostly on methods developed by Rheinhardt *et al.* (2002) for similar wetlands along the Atlantic and Gulf of Mexico Coastal Plains.

After Reference Wetland sampling was completed, the Principal Investigators as well as local, regional, and national experts in hydrology, soil sciences, ecology, and biology evaluated the data at workshops. The primary objective of the workshops was to select and scale variables for use in field assessments of wetlands in the second phase of the project, the *Assessment Phase*. Variables listed in bold in Table 1 are the variables that were selected for use in calculating FCI scores for the HGM models listed in Table 2. Table 2 shows how variables were combined to calculate Functional Capacity Scores (FCI) for five HGM models for the riverine wetland class and four HGM models for the flats wetland class.

The HGM models are chosen to represent broad categories of ecological processes in wetland ecosystems. The hydrology function is found in all HGM models because of the importance of hydrologic conditions in wetlands. The variables that are used to evaluate the hydrology function typically are chosen to represent physical features (e.g., stream condition, the presence of absence of human alterations to the stream, the presence of drainage features in the wetland) that would result in alterations of the site water balance. The biogeochemical function is representative of nutrient cycling processes that occur in wetlands. Because it is not possible to measure rates of nutrient cycling in short-term wetland assessments, the biogeochemistry models incorporates structural features of the wetland system that are important elements of nutrient cycling (e.g., the presence of mature vegetation that includes both living and dead biomass). The plant community and habitat functions are representative of the biodiversity and structural features of wetlands. The models typically include variables that quantify features of the vegetation including biomass and species composition. The habitat model usually represents features of the vegetation that provide habitat for animals. The landscape function is usually chosen to represent the condition of the landscape adjacent to the assessment site. This model is important because the characteristics of the adjacent landscape determine the degree to which the assessment site may be impacted by human activities.

As indicated, variables were scaled from 0.0 to 1.0 and HGM models were mathematically organized to calculate FCI scores, that ranged from 0.0 to 1.0. A score of 1.0 means that the function at a site is in a condition equivalent to a reference standard site (i.e., the least altered functionality). As the FCI score declines, the condition of the wetland function degrades until the function is absent at a score of 0.0. Brinson *et al.* (1995), Smith *et al.* (1995), Whigham *et al.* (1999) and Rheinhardt *et al.* (2002) provide more detailed description of procedures used to scale HGM variables and develop HGM models to calculate FCI scores.

During the *Assessment Phase* of the project, sites were chosen using protocols developed by the U.S. Environmental Protection Agency's Environmental Monitoring and Assessment Program (EMAP). One of the PIs (DEW) provided EMAP staff with the most recent digital wetland maps for the Nanticoke River watershed. A Generalized Random Tessellation Stratified (GRTS) design (Stevens and Olsen 1999, 2000) was used to draw the sample from the maps and generate potential sample sites identified by latitude and longitude. The basic concept of GRTS design is to construct a random spatial stratification using equal-sized tessellation cells, and then to select a point at random within each cell. A spatial address is constructed using the pattern of subdivision so that the result is a spatially well-distributed sample. The final set of assessment sites is well-dispersed over the accessible portion of the population (Stevens and Olsen, in review, 2002) and each point will have a known probability of being selected.

Potential sites were chosen for inclusion in the set of assessment sites only when it had been determined that they were actually wetlands of the targeted class (flat or river-

HGM function	Equation used to calculate FCI score
Flats subclass	
Hydrology	$0.25*V_{FILL} + 0.75*V_{DRAIN}$
Biogeochemistry	$((V_{MICRO} + (V_{SNAG} + V_{TBA} + V_{TDEN})/3)/2) *$ Hydrology FCI
Habitat	$(V_{DISTUR} + ((V_{TBA} + V_{TDEN})/2) + V_{SHRUB} + V_{SNAG})/4$
Plant Community	$((V_{TREE} + V_{HERB})/2) * V_{RUBUS}$
Riverine subclass	
Hydrology	$SQRT((V_{STREAMIN} + (2 * V_{FLOODPLAIN}))/3) * V_{STREAMOUT})$
Biogeochemistry	$(V_{TBA} +$ Hydrology FCI$)/2$
Habitat	$(((((V_{TBA} + V_{TDEN})/2) + V_{SHRUB} + V_{DISTURB})/3) + V_{STREAMIN})/2$
Plant Community	$(.75 * ((V_{TREE} + V_{SAPLING})/2)) + (.25 * ((V_{VINE} + V_{INVASIVE})/2))$
Landscape	$(.5 * V_{NEARBUFFER}) + (.25 * V_{FARBUFFER}) + (.25 * V_{STREAMOUT})$

Table 2. HGM models used to calculate functional capacity index (FCI) scores for riverine and flats wetland classes. Variables are listed and described in Table 1.

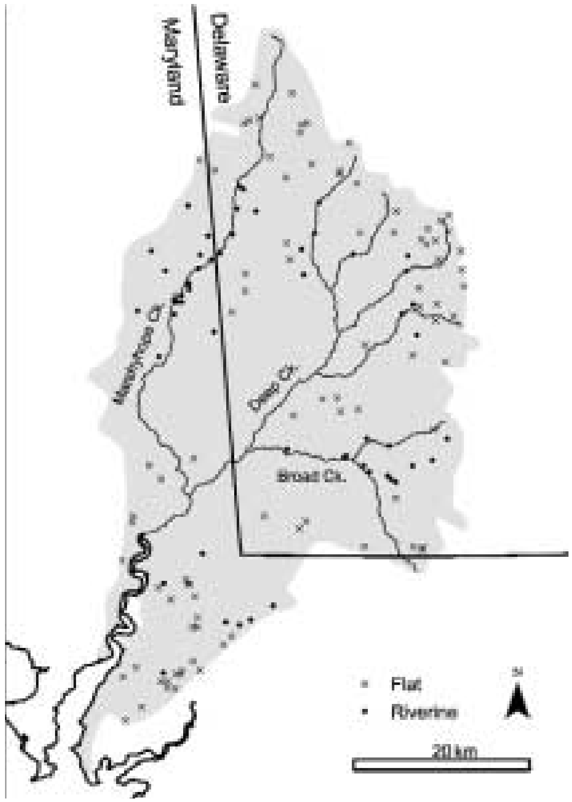


Figure 4. Location of assessment sites in the Nanticoke River watershed for riverine and flats subclasses.

ine) and permission for access had been obtained. Figure 4 shows the distribution of assessment sites for both classes of wetlands. The first 17 flats and 15 riverine sites that met our criteria and to which we were allowed access were used as sites for testing the final protocols and models. Following the field testing, final versions of the data sheets and variable scaling procedures were prepared for use in the *Assessment Phase*.

Field-assessments were conducted by teams under the supervision of one of the authors (ADJ). The field teams received training from two of the authors (DFW, ADJ) and they followed formal quality assurance and quality control procedures (The Nature Conservancy, 2000; Whigham *et al.*, 2000). Assessment teams consisted of individuals hired for the project and volunteers, mostly provided through contacts with The Nature Conservancy.

Data compiled during the assessment phase of the project were scanned from the field datasheets to create computer files using procedures developed by EMAP under the supervision of one of the authors (MEK). Electronic data files were checked with field data sheets and corrected.

Comparison of assessment data with remotely sensed spatial data

One of our objectives was to determine if it would be possible to use remotely acquired spatial data to produce site assessments with an acceptable degree of accuracy. We evaluated a variety of mapped spatial data (Table 3) for their potential to predict wetland conditions as assessed by HGM field-based assessments. In this paper, we focus on preliminary results using land cover data (Table 3) and metrics of stream disturbance status (natural, channelized, or artificial ditch; Tiner *et al.*, 2000, 2001). For each wetland, land cover proportions and lengths of excavated and natural stream channels were determined for radial distances of 100, 500 and 1000 meters from the sampling point provided by EMAP. Step-wise multiple regression analysis was used to determine the relationship between the independent variables and the measured HGM variables (Table 1) and FCI scores (Table 2) for riverine and flats subclasses.

Results

Selection of assessment sites

Digital wetland maps were used to evaluate up to 1050 potential assessment sites from a list of 1,992 random points provided by EMAP. Based on an interpretation of digital maps of the 1050 potential sites, we selected a subset of 455 sites to which we sought access. Sites were examined

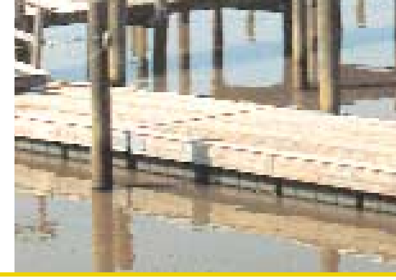


Table 3. Spatial data sets with sources or contacts.

Data set	Source
Orthophotography for Maryland	http://www.dnr.state.md.us/MSGIC/techtool/samples/metadata/doqq.htm
Orthophotography for Delaware	http://bluehen.ags.udel.edu/spatlab/doqs/_doq.html
EPA EMAP land cover	U.S. EPA., 1994
NLCD land cover	Vogelman *et al.*, 2001SSURAGO NRCS county soils data http://www.ftw.nrcs.usda.gov/ssur_data.html ftp://ftp.ftw.nrcs.usda.gov/pub/ssurgo/online98/data/ http://bluehen.ags.udel.edu/spatlab/soils/
EPA Reach File 3 stream maps	http://www.epa.gov/r02earth/gis/atlas/rf3_t.htm
US Census TIGER road files	http://www.census.gov/geo/www/tiger/index.html
Stream maps classified by disturbance	Tiner *et al.*, 2000; Ralph Tiner (unpublished data)

in the order provided by EMAP. The coding associated with existing digital wetland maps could not be used to determine the hydrogeomorphic classification of individual wetlands. Subsequently, each potential wetland assessment site identified by EMAP had to be visited to evaluate the following criteria, which all had to be met in order for a site to be selected:

- Point was in the respective testing or assessment group specified by EMAP
- Point was in the Nanticoke River watershed
- Point was a wetland
- Point was in a non-tidal wetland
- Point was in a wetland in the flats or riverine HGM subclass
- Point was not in a farmed wetland
- Landowner permission had been granted to conduct the assessment

One of the most time consuming aspects of this part of the project was the process of obtaining permission from private landowners to visit potential assessment sites. First, landowners were identified through the use of public ownership documents. We then examined the lists of owners and identified individuals who would be willing to attempt to communicate with the landowner by calling or scheduling a meeting. We received no response from 38% of the contacts and 17% of the contacts denied access. We gained permission to sample 201 sites. Once contact had been made with landowners, we obtained access to all of the publicly owned sites and 67% of the privately owned sites. Contacting landowners, follow-up contacts with landowners, and examination of the sites to determine if they would be included in the study took approximately 168 person-days (1,200 hours). For comparison, two other major components of the study took less time. Site selection and forming and training field crews took 97 person-days (776 hours). Sampling assessment sites required 145 person days (1160 hours).

Assessment sites for both wetland classes were distributed across the entire watershed (Figure 4) but there was a bias toward public sites in the riverine subclass (D. Stevens, personal communication). The bias was most likely the result of a lower level of accessibility to privately owned riverine sites. EMAP staff will be conducting further tests to determine if adjustments need to be made in the final interpretation of the assessment data.

Range of variability of FCI scores

A goal of any HGM protocol is to select variables that quantitatively express the range of natural variation

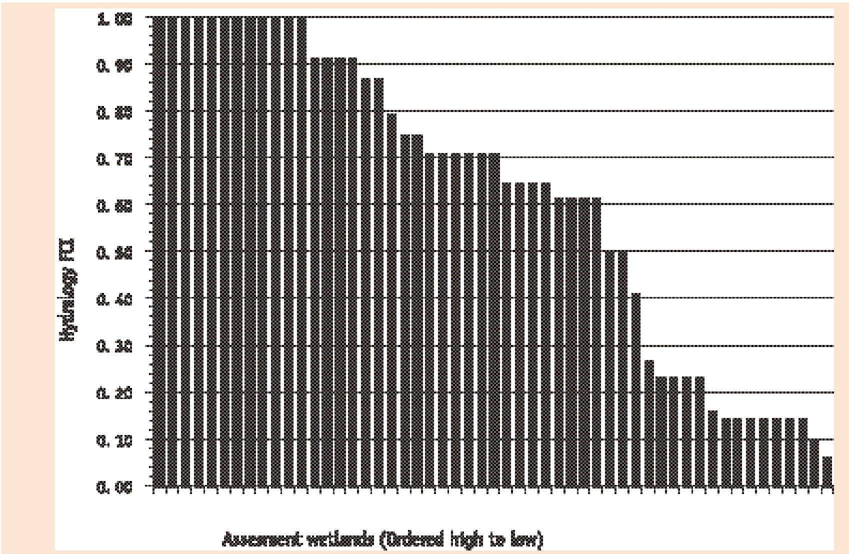


Figure 5. Distribution of FCI scores for the hydrology function for the riverine subclass sampled in the Nanticoke River watershed. Sites are aligned so that FCI scores vary from high (left) to low (right). The hydrology model for the subclass is provided in Table 2.

across the set of reference sites (Brinson & Rheinhardt, 1996; Wakeley & Smith, 2001). In this project, approximately half of the variables that were initially chosen were eventually used in the HGM models (Tables 1 and 2). FCI scores shown in Figure 5 are typical of scores for all of the models in both hydrogeomorphic subclasses. FCI scores varied from 1.0 (reference standard conditions with no detectable impacts) to 0.1 (function present but at a very low level). These results suggest that the majority of the wetlands in the two classes have been degraded from reference standard conditions. Only a small percentage of un-impacted wetlands remain (e.g., sites with FCI scores > 0.90 for all functions), suggesting that there is a high potential for restoration of wetland functions within the watershed. Further analysis of the FCI scores and variable scores will be conducted to determine which variables were most responsible for lower FCI scores at impacted sites and which wetland features need to be considered in the development of restoration goals.

In addition, we will be conducting further analyses to evaluate how wetland condition varies spatially throughout the watershed. Locations of streams (Marshyhope Creek, Deep Creek, Broad Creek) that drain three subwatersheds are shown on Figure 3. Table 4 shows mean FCI scores for the five riverine functions for Marshyhope Creek, Deep Creek and Broad Creek subwatersheds. Mean FCI scores were significantly lower for four of the functions (hydrology, biogeochemistry, habitat, and landscape) in the Deep Creek subwatershed (Table 4). Spatial information of this type can potentially be used to identify problem areas within the watershed as well as targeting areas within the watershed for restoration. Analysis of spatial information will also allow us to further evaluate the adequacy of the site selection process. The ratio of public to privately owned assessment sites was lower in the Deep Creek subwatershed, potentially resulting in a bias toward lower quality private sites with lower FCI scores.

Suitability of using geographic data to assess individual wetland sites

Use of the mapped digital data to predict HGM functions produced variable results. For the flats subclass, there were significant stepwise multiple regressions for each of the HGM functions (data not shown) and the regressions

Table 4. Mean FCI scores for five HGM functions for the riverine subclass for the three large subwatersheds in the Nanticoke River system. The number of riverine assessment sites in each subwatersheds were: Marshyhope = 24, Deep Creek = 10, and Broad Creek = 13. For each function, means that differ for the subwatersheds have different superscripts.

Subwatershed	Hydrology	Biogeochemistry	PlantCommunity	Habitat	Landscape
Marshyhope Creek	.701[a]	.772[a]	.947[a]	.859[a]	.788[a]
Deep Creek	.236[b]	.495[b]	.807[a]	.431[b]	.584[b]
Broad Creek	.683[a]	.759[a]	.809[a]	.727[a]	.770[a]

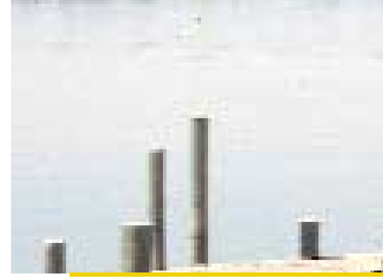


Table 5. Stepwise multiple regression results for riverine HGM functions (dependent variables) and landscape cover data (independent variables). All models shown in the Table were significant at $p < 0.0001$. The sign (+/-) in front indicates whether the variable is positively or negatively related to the HGM function.

HGM Function	No. of Variables	Variables	R2
Biogeochemistry	3	-ex100 + nat1000 – DEV100	0.51
Habitat	2	-ex100 + nat1000	0.42
Hydrology	5	-ex100 + nat1000 +FOREST100 +FOREST1000 –FORDEC100	0.70
Landscape	6	-ex100 –ex1000 +nat1000 –CROP100 –DEV1000 +FOREVER1000	0.70
Plant Community	2	-ex500 –DEV100	0.31

Variable names are:

ex100	Length of excavated stream channel (ditches and channelized) in 100 m circle around the sample point.
ex500	Length of excavated stream channel (ditches and channelized) in 500 m circle around the sample point.
ex1000	Length of excavated stream channel (ditches and channelized) in 1000 m circle around the sample point.
nat1000	Length of natural stream channel in 1000 m circle around sample point.
DEV100	Proportion of total developed land (low + high intensity development in 100 m circle around the sample point).
DEV1000	Proportion of total developed land (low + high intensity development in 1000 m circle around the sample point).
FOREST100	Total amount of forest within 100 m of the sample point.
FOREST1000	Total amount of forest within 1000 m of the sample point.
FORDEC100	Total amount of deciduous forest within 100 m of the sample point.
FOREVER1000	Total amount of evergreen forest within 1000 m of the sample point.
CROP100	Total amount of crop within 100 m of the sample point.

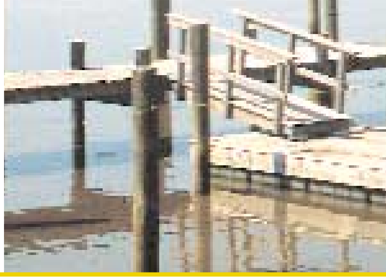
explained between 17 and 44% of the variability. Multiple regressions were more successful in predicting FCI scores for the riverine class than the flats class (Table 5). All of the multiple regressions in Table 5 were significant at $p < 0.0001$ and they accounted for between 31% and 70% of the variation in the FCI scores. One variable (length of excavated stream channel within 100 or 500 meters of the site where the assessment was conducted) had a negative relationship to the FCI scores for all models. This result clearly suggests that channelization results in effective drainage of sites and has a negative impact on wetland function as measured by HGM scores. Land-use categories were also important. Increasing amounts of developed land and crop land near the assessment site had a negative influence on FCI scores and the greater the amount of forested land near the site, the higher the FCI score. These results suggest that individual wetlands have important linkages to adjacent land uses and that degradation of areas adjacent to wetlands results in negative impacts of ecological functions in the wetlands.

Discussion

As described earlier, the project was divided into a *Development Phase* and an *Assessment Phase*, with each phase taking approximately one year to complete. We believe that the two phases are equally important to overall success of a project. The *Development Phase* is essential if site-specific assessments are to be conducted in the second phase. The selection and sampling of reference sites and the selection and scaling of variables are essential elements of any field-based HGM assessment. The necessity of selecting reference sites that represent the range of condition for a given wetland class has been described by Brinson and Rheinhardt (1996). Data from reference sites are essential in the selection of HGM variables that can be used to quantify differences between assessment sites. Both selection and sampling of sites during the *Development Phase* require adequate training of field teams (Whigham *et al.* 1999), implementation of procedures to assure accuracy of data gathering and reporting, and development of standard methods for collecting field data (Wakeley & Smith, 2001). The reader can refer to several HGM guidebooks to learn more about the procedures that have been suggested for selecting HGM variables and for selecting and sampling reference wetland sites using HGM procedures (Adamus & Field, 2001; Hauer *et al.*, 2002; Rheinhardt *et al.*, 2002). The *Development Phase* is time consuming and costly; thus it is often cited as one reason why the HGM approach to wetland assessment has not been used more widely. While it is unfortunate that there are no faster ways to complete the

Development Phase, the results are worth the effort because field-assessments can be done in less than one day when field-tested protocols have been developed. In addition, once the procedures have been developed and verified, methods can be applied in many locations. Thus, the product of the investment in the *Development Phase* has applications beyond the initial assessment and the potential for continued use in a monitoring and assessment program that supports decision making.

While we have not reached any final conclusions regarding the ecological condition of wetlands in the watershed, the approach that we have used clearly suggests that there is a wide range of conditions in the watershed and that most wetlands in the watershed have been degraded at some level. Preliminary data further suggest that wetland condition differs among wetlands in different subwatersheds of the Nanticoke basin. Finally, the use of spatial geographic data can be important in assessing wetland condition at the scale of entire watersheds for several reasons. First, spatial data can be effectively used to identify and conduct preliminary interpretations of potential assessment sites. Second, spatial data at appropriate levels of resolution can provide input variables to HGM models. Third, mapped spatial data has the potential to be used as a surrogate for field-based assessments when properly calibrated with field assessments. This study will provide useful information for designing future watershed-based assessments that employ a combination of field-based sampling and assessment based on spatial data.



Acknowledgments

The research was funded in part by the U.S. Environmental Protection Agency under cooperative agreement 82681701. It has been reviewed by the National Health and Environmental Effects Research Laboratory's Western Ecology Division and approved for publication. Approval does not signify that the contents reflect the views of the Agency, nor does mention of trade names or commercial products constitute endorsement or recommendation for use. Additional funding was provided by the Smithsonian Institution in the form of fellowships to a Post-doctoral fellow (Vladimir Samarkin) and student internships through the Work-Learn Program. The authors acknowledge the assistance of the following individuals who helped with various parts of the project: Chris Bason, Stephanie Behles, Jeff Lin, Mary Pittek, Marcia Snyder, Arthur Spingarn, Don Stevens, Richard Sumner, Ryan Szuch, Christine Whitcraft, Mike Yarkcusko and many dedicated volunteers who assisted with field data collection.

Abstract

Ecological processes in wetlands result in important societal values, whether one is considering an individual wetland or all of the wetlands within a catchment (watershed). In addition to providing habitats for numerous species, wetlands typically intercept surface and groundwater and improve water quality by removing nutrients, contaminants, and sediments. A variety of approaches have been developed to assess the ecological condition of individual wetlands, but less progress has been made in developing approaches to evaluating the ecological condition of wetlands at the scale of entire watersheds. In this paper we describe an approach to assessing the ecological condition of two classes of wetlands in the Nanticoke River watershed, a subwatershed in the Chesapeake Bay drainage of North America. We used the hydrogeomorphic (HGM) approach to assess the ecological condition of wetlands along nontidal streams (riverine class) and wetlands associated with poorly drained soils on interfluves (flats class). Sampling protocols developed by the U.S. Environmental Protection Agency's Environmental Assessment and Monitoring Program were used to select a spatially unbiased sample of sites for field-based assessments. Statistical procedures were used to determine the relationships between data compiled in the field-based assessments and spatial data from remote sensing or other mapping efforts. We wanted to determine if available geographic data could be used to assess individual wetlands or the overall condition of wetlands in the watershed without having to do site-specific assessments based on field sampling. The HGM approach to wetlands assessment appears to be a useful methodology when it is applied in combination with a spatially unbiased method for selecting sampling sites. There were significant relationships between results of HGM assessments and mapped geographic data, but the strengths of the relationships were variable, demonstrating potential limitations to the use of mapped geographic data to assess wetlands condition in relatively flat landscapes such as those present in the Nanticoke River watershed. Future improvements in the resolution of GIS data, however, should result in better correlations between GIS-based assessments and field-based assessments of wetlands.

References

Adamus, P. R. & D. Field, 2001. Guidebook for hydrogeomorphic (HGM) – based assessment of Oregon wetland and riparian sites. I. Willamette Valley Ecoregon, riverine impounding and slope/flats sub-classes. Volume IA: Assessment methods. Oregon Division of State Lands, Salem, OR.

Bartoldus, C. C., 1999. A Comprehensive review of Wetland Assessment Procedures: A Guide for Wetland Practitioners. Environmental Concern Inc., St. Michaels, MD, USA.

Bohlen, C. C. & R. Friday, 1997. Riparian and terrestrial issues in the Chesapeake: A landscape management perspective. In: Simpson & Christensen (eds.) Ecosystem function and human activities. Reconciling economics and ecology. Chapman & Hall: 95-125.

Brinson, M. M. & R. D. Rheinhardt, 1996. The role of reference wetlands in functional assessment and mitigation. Ecological Applications 6: 69-76.

Brinson, M. M., R. F. Hauer, L. C. Lee, W. L. Nutter, R. D. Rheinhardt, R. D. Smith & D. F. Whigham, 1995. Guidebook for application of hydrogeomorphic assessments to riverine wetlands. Army Corps of Engineers, Waterwas Experiment Station, Vicksburg, MS, USA. Technical Report TR-WRP-DE-11.

Cedfeld, P. T., M. C. Watzin & B. D. Richardson, 2000. Using GIS to identify functionally significant wetlands in the Northeastern United States. Environmental Management 26: 13-24.

Cormier, S. M., M. Smith, S. Norton & T. Neiheisel, 2000. Assessing ecological risk in watersheds: A case study of problem formulation in the Big Darby Creek watershed, Ohio, USA. Environmental Toxicology and Chemistry 19: 1082-1096.

Crumpton, W. G., 2001. Using wetlands for water quality improvement in agricultural watersheds; the importance of a watershed scale approach. Water Science and Technology 44: 559-564.

Detenbeck, N. E., S. L. Batterman, V. J. Brady, J. C. Brazner, V. M. Snarski, D. L. Taylor, J. A. Thompson & J. W. Arthur, 2000. A test of watershed classification sytems for ecological risk assessment. Environment, Toxicology and Chemistry 19: 1174-1181.

Dorioz, J. M. & A. Ferhi, 1994. Non-point pollution and management of agricultural areas: Phosphorus and nitrogen transfer in an agricultural watershed. Water Research 28: 395-410.

Greiner, M. & C. Hershner, 1998. Analysis of wetland total phosphorus retention and watershed structure. Wetlands 18: 142-149.

Hauer, F. R., B. J. Cook, M. C. Gilbert, E. J. Clairain Jr. & R. D. Smith, 2002. A Regional Guidebook for Applying the Hydrogeomorphic Approach to Assessing Wetland Functions of Riverine Floodplains in the Northern Rocky Mountains. ERDC/EL TR-02-21. U. S. Army Engineer Research and Development Center, Vicksburg, MS.

Johnston, C. A., N. E. Detenbeck & G. J. Niemi, 1990. The cumulative effect of wetlands on stream water quality and quantity. A landscape approach. Biogeochemistry 10: 105-142.

Jordan, T. E., D. L. Correll & D. E. Weller, 1997. Effects of agriculture on discharges of nutrients from Coastal Plain watersheds of Chesapeake Bay. Journal of Environmental Quality 26: 263-284.

Kuusemets, V. & U. Mander, 1999. Ecotechnological measures to control nutrient losses from catchemetns. Water Science and Technology 40: 195-202.

Leibowitz, S. G., C. Loehle, B-L, Li & E. M. Preston, 2000. Modeling landscape functions and effects: a network approach. Ecological Modelling 132: 77-94.

Lemly, A. D., 1997. Risk assessment as an environmental management tool: Considerations for freshwater wetlands. Environmental Management 21: 343-358.

O'Neill, M. P., J. C. Schmidt, J. P. Dobrowolski, C. P. Hawkins, & C. M. Neale, 1997. Identifying sites for riparian wetland restoration: Application of a model to the Upper Arkansas River basin. Restoration Ecology 5: 85-102.

Phillips, P. J., J. M. Denver, R. J. Shedlock & P. A. Hamilton, 1993. Effect of forested wetlands on nitrate concentrations in groundwater and surface water on the Delmarva Peninsula. Wetlands 13: 75-83.

Reed, T. & S. R. Carpenter, 2002. Comparison of P-yield, riparian buffer strips, and land cover in six agricultural watersheds. Ecosystems 5: 568-577.

Rheinhardt, R. D., M. C. Rheinhardt, & M. M. Brinson. 2002. A Regional Guidebook for Applying the Hydrogeomorphic Approach to Wet Pine Flats on Mineral Soils in the Atlantic and Gulf Coastal Plains. U. S. Army Corps of Engineers, Waterways Experiment Station, Vicksburg, MS.

Smith, R. D., A Ammann, C. Bartoldus & M. M. Brinson, 1995. An approach for assessing wetland functions using hydrogeomorphic classification, reference wetlands, and functional indices. U. S. Army Corps of Engineers, Waterways Experiment Station, Vicksburg, MS, USA. Technical Report TR-WRP-DE-10.

Stevens, Jr., D. L. & A. R. Olsen, 1999. Spatially Restricted Surveys Over Time for Aquatic Resources. Journal of Agricultural, Biological, and Environmental Statistics 4:415-428.

Stevens, Jr., D.L., & A. R. Olsen, 2000. Spatially-restricted Random Sampling Designs for Design-based and Model-based Estimation. In: Accuracy 2000: Proceedings of the 4th International Symposium on Spatial Accuracy Assessment in Natural Resources and Environmental Sciences. Delft University Press: 609-616.

Stevens, Jr., D.L., & A. R. Olsen., In review, 2002. Spatially-Balanced Sampling of Natural Resources in the Presence of Frame Imperfections. Submitted to Journal of the American Statistical Association.

The Nature Conservancy, 1994. Nanticoke-Blackwater River Bioreserve Strategic Plan. Draft 6-10-94. Maryland Nature Conservancy, Chevy Chase, MD.

The Nature Conservancy, 2000. Nanticoke wetland monitoring study field operations. Plan year 2000 field season. Order No: 9D-1186-NTNX. Delaware Chapter of The Nature Conservancy, Wilmington, DE.

Tiner, R. W., 1985. Wetlands of Delaware. U. S. Fish and Wildlife Service, National Wetlands Inventory, Newton Corner, MA and Delaware Department of Natural Resources and Environmetntal Control, Wetlands Section, Dover, DE. Cooperative Publication.

Tiner, R. W. & D. G. Burke, 1995. Wetlands of Maryland. U. S. Fish and Wildlife Service, Ecological Services, Region 5, Hadley, MA and Maryland Department of Natural Resources, Annapolis, MD. Cooperative Publication.

Tiner, R., M. Starr, H. Berquist & J. Swords, 2000. Watershed-based wetland characterization for Maryland's Nanticoke River and Coastal Bays watersheds: A preliminary assessment report. U. S. Fish and Wildlife Service, Hadley, MA.

Tiner, R., H. Berquist, J. Swords & B. J. McClain, 2001. Watershed-based wetland characterization for Delaware's Nanticoke River and Coastal Bays watersheds: A preliminary assessment report. U. S. Fish and Wildlife Service, Hadley, MA.

U. S. Environmental Protection Agency, 1994. Chesapeake Bay water-shed pilot project. BAP620R94020. USEPA EMAP Center, Research Triangle Park, NC.

Vogelmann, J. E., S. M. Howard, L. Yang, C. R. Larson, B. K. Wulie & N. Van Driel, 2001. Completion of the 1990s national land cover data set for the conterminous United States from landsat thematic mapper data and ancillary data sources. Photogrammetric Engineering and Remote Sensing 67: 650-652.

Wakeley, J. S. & R. D. Smith, 2001. Hydrogeomorphic Approach to Assessing Wetland Functions: Guidelines for Developing Regional Guidebooks - Chapter 7 Verifying, Field Testing, and Validating Assessment Models, ERD/EL TR-01-31, U.S. Army Engineer Research and Development Center, Vicksburg, MS.

Weller, C. M., M. C. Watzin & D. Wang, 1996. Role of wetlands in reducing phosphorus loading to surface water in eight watersheds in Lake Champlain Basin. Environmental Management 20: 731-739.

Whigham, D. F., C. Chitterling & B. Palmer, 1988. Impacts of fresh-water wetlands on water quality: A landscape perspective. Environmental Management 12: 663-671.

Whigham, D. F., L. C. Lee, M. M. Brinson, R. D. Rheinhardt, M. C. Rains, J. A. Mason, H. Khan, M. B. Ruhlman & W. L. Nutter, 1999. Hydrogeomorphic (HGM) assessment – a test of user consistency. Wetlands 19: 560-569.

Whigham, D. F., D. E. Weller & T. E. Jordan, 2000. Quality assurance project plan for Nanticoke wetland assessment study. U. S. Environmental Protection Agency, Environmental Monitoring and Assessment Program, Corvallis, OR. Internal Report, June 2000.


Foto's: Saxifraga, Ben Delbaere

Landscape planning
Nitrogen budget
Riparian buffer zones
Spatial scale
Territorial ecological networks

Scaling in territorial ecological networks

Territorial ecological networks are coherent assemblages of areas representing natural and semi-natural landscape elements that need to be conserved, managed or, where appropriate, enriched or restored in order to ensure the favourable conservation status of ecosystems, habitats, species and landscapes of regional importance across their traditional range (Bennett, 1998). In this study we demonstrate the hierarchical character of territorial ecological networks, recognize common elements and functional differences between hierarchical levels, and analyze the downscaling and upscaling of the functions of ecological networks. Emerging from the examples of ecological networks at different hierarchical levels, we highlighted following common principles: connectivity, multifunctionality, continuity, and plenipotentiality.

ÜLO MANDER, MART KÜLVIK & ROBERT JONGMAN

Prof. Ü. Mander, Institute of Geography, University of Tartu, Vanemuise 46
51014, Tartu, Estonia,
mander@ut.ee
Dr. M. Külvik, Environmental Protection Institute, Estonian Agricultural University, POB 222, Tartu, 50002, Estonia,
mkulvik@envinst.ee
Dr. R.H.G. Jongman, Alterra, Green World Research, PO-box 47 6700 AA, Wageningen, The Netherlands,
r.h.g.jongman@alterra.wag-ur.nl

Landscape ecology provides the insight that nature at a landscape level is a relatively dynamic system reacting to a complex of environmental and land use conditions. It has been declared that landscape represents a crucial organizational level and special scale, at which both the effects of global change, as well as site-based biodiversity trends, are apparent, hence, at which appropriate responses will need to be implemented (Hobbs, 1997). The meaningful way in which humans interpret this nature at a landscape scale, and as a modelling instrument in spatial or physical planning, can be called an ecological network (Cook & van Lier, 1994). Most specific initiatives to develop ecological networks meet and suit the specific circumstances evident in the particular geographic and, even more importantly, hierarchical context.

The widely used European-level approach considers territorial ecological networks as coherent assemblages of areas representing natural and semi-natural landscape elements that need to be conserved, managed or, where appropriate, enriched or restored in order to ensure the favourable conservation status of ecosystems, habitats, species and landscapes of regional importance across their traditional range (Bennett, 1998).

In addition to this approach, there are a wide range of names worldwide given to such 'patch and corridor' spatial concepts: greenways in the USA, Australia and New Zealand (Ahern, 1995; Hobbs, 1997; Viles and Rosier, 2001), ecological infrastructure, ecological framework (van Buuren and Kerkstra, 1993), extensive open space systems, multiple use nodules, wildlife corridors, landscape restoration network (Ahern, 1995), habitat networks, territorial systems of ecological stability, framework of landscape stability (Jongman, 1995). In Estonia, a concept of 'the network of ecologically compensating areas' (Mander *et al.*, 1988) has been developed since the early 1980s. This network can be observed as a landscape´s subsystem – an ecological infrastructure – that counterbalances the impact of the anthropogenic infrastructure in the landscape. In comparison with the traditional biodiversity-targeted approach, this concept also considers the material and energy cycling, socio-economic and socio-cultural aspects.

The network of ecologically compensating areas is, like all territorial ecological networks, a multilevel hierarchical system. Their hierarchy emerges from both the spatial range and functions. Although ecological networks are already widely used practice in landscape/territorial planning and nature conservation (Cook and Van Lier, 1994; Ahern, 1995; Jongman, 1995; Bouwma *et al.*, 2002), there are few works available on the hierarchical analysis of territorial ecological networks (Cook, 2002; Villeumier & Prelaz-Droux, 2002).

The main objectives of this study are: (1) to demonstrate the hierarchical character of territorial ecological networks, (2) to recognize common elements and function-

al differences between hierarchical levels of territorial ecological networks; (3) to analyze the downscaling and upscaling of the functions of ecological networks and their spatial distribution.

Considering hierarchy in the application of the ecological network model in practice helps to reflect the complexity of pattern and processes at the landscape level. One of the ways to downscale the functions of an ecological network is to use a strategy based on suitability criteria. This approach helps to reveal, evaluate and exploit the impact of protected and sparsely populated areas on the environment in the broader sense. Likewise, it has been used to identify and measure the suitability of potential sites for ecological network development in residential areas (Miller *et al.*, 1998). As an example, a GIS-based habitat suitability analysis for the designing of national-level ecological networks in Estonia is presented in this paper. For the upscaling approach from the micro-scale ecological network to the meso- and macro-scale level, a nutrient fluxes modeling attempt in riparian buffer zones will be presented. The use of point models step-by-step within elementary watersheds helps to describe the changing gradient of nutrient fluxes along the water filtration path and allows the creation of bridges between the different hierarchical levels of ecological networks.

Roots of the concept

Development of the idea of territorial ecological networks may be largely based on the central place theory elaborated by J.H. von Thünen (1826, 1990), W. Christaller (1933, 1966) and A. Lösch (1954). Enhanced by the Von-Thünen-Christaller-Lösch theory of central places and their hierarchy, Rodoman (1974) used the idea of influence pattern and spatial hierarchy to advance the concept of polarized landscapes. According to this approach, two main poles – centres of human activities (e.g., cities) on the one hand, and centres of pristine (undisturbed) nature (e.g., large forest and swamp areas) on the other hand – create the hierarchical gradient fields of interactions. Thus, it allows the use of the Von Thünen-Christaller-Lösch model for reverse situations, not proceeding from the development of economic but ecological benefit. In this case ecological benefit means first of all less disturbance by human activities (Külvik *et al.*, 2003).

Structural components as indicators of functional hierarchy

A network of ecologically compensating areas is a functionally hierarchical system with the following components: (A) core areas, (B) corridors; functional linkages between the ecosystems or resource habitat of a species enabling the dispersal and migration of species and resulting in a favourable effect on genetic exchange (individuals, seeds, genes) as well as on other interactions between ecosystems; corridors may be continuous (linear;

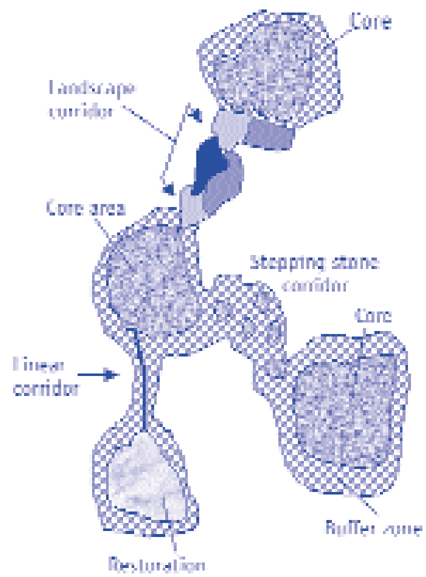


Figure 1. Schematic example of an ecological network (from Bouwma *et al.*, 2002; with permission of ECNC and I. Bouwma).

Saunders *et al.*, 1991), interrupted (stepping-stones; Brooker *et al.*, 1999) and/or landscape corridors (scenic and valuable cultural landscapes between core areas), (C) buffer zones of core areas and corridors, which support and protect the network from adverse external influences, and (D) nature development and/or restoration areas that support resources, habitats and species (Bennett, 1998; Bouwma *et al.*, 2002; Figure 1).

Corridors which provide connectivity between the core areas can be considered as key elements of ecological networks. According to Ahern (1995), ecological corridors and greenways are a linked or spatially-integrated network of lands that are owned or managed for public uses including biodiversity, scenic quality, recreation and traditional agriculture. The viability of certain processes in landscapes is dependent on connectivity (the movement of wildlife species and populations, the flow of water, the flux of nutrients, and human movement). Without connectivity, these processes and functions may not otherwise occur. However, connectivity must be understood in terms of the process or function that it is intended to support.

Movement, which assumes connectivity, is itself the product of evolutionary pressures contributing in many ways to the survival and the reproduction of the animal. Animals move through their home range, but may also move long distances from where they were born and their kin remain. Three kinds of movements can be distinguished (Caughley & Sinclair, 1994):

Local movements- these are movements within a home range and are on smaller scales;

- Dispersal- movement from the place of birth to the site of reproduction, often away from its family group and usually without return to place of birth;
- Migration- movement back and forth on a regular basis, usually seasonally, e.g. from summer range to winter range to summer range.

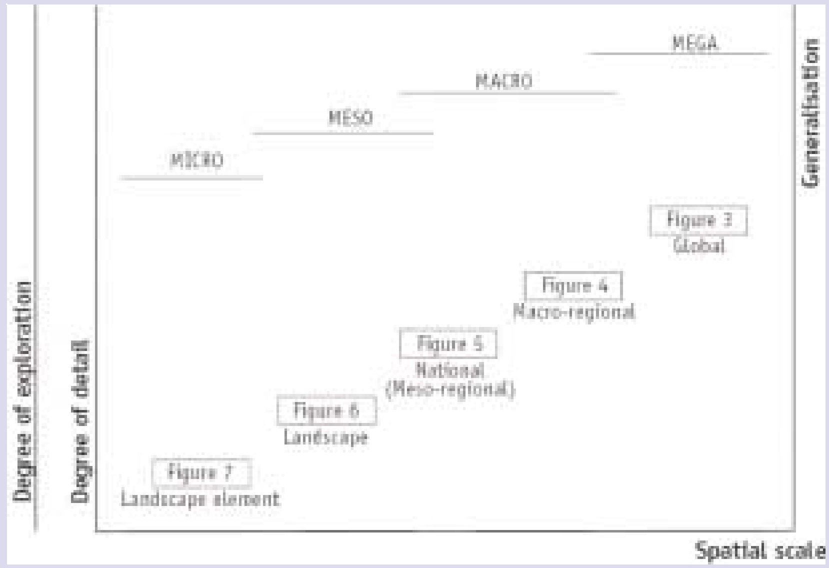


Figure 2. Hierarchy levels of ecological networks and according representative figures of this paper. The degree of detail and the exploredness are increasing and generalization is decreasing towards lower (detail) levels.

Local movements, within the home range of a species for foraging, hiding from enemies and optimizing living conditions, are normally not included in the analyses and implementation of ecological network. However, this kind of movement is most important at lower spatial scales of ecological networks.

Spatial hierarchy

Most specific initiatives to develop ecological networks – either theoretically or in practice – consider the specific circumstances evident in the particular hierarchical context. The most practicable is the approach that proceeds from the traditional scaling of maps in cartography: 1:500; 1:1000; 1:5000; 1:10,000; 1:50,000; 1:100,000, 1:500,000 etc. Mander *et al.* (1995) intuitively defines the network components at four levels: (a) mega-scale: large natural core areas (>10,000 km²) and their buffer zones, sometimes connected with corridors; (b) macro-scale: large natural core areas (>1000 km²) surrounded by buffer

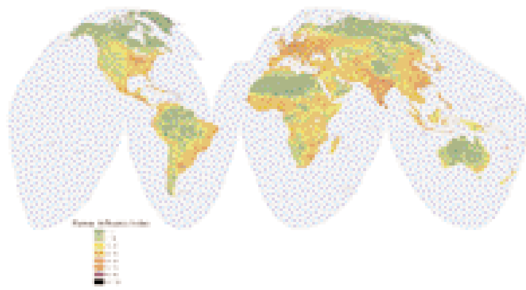


Figure 3. The map of the Human Footprint as a basis for the ecological network system at the global scale (Sanderson *et al.*, 2002). Summarized factors of anthropogenic pressure have been used, such as the Human Influence Index, which is the quantitative basis for the map. Adopted from www.ciesin.columbia.edu/wild_areas/. The full list of biomes is available at *www.wcs.org/humanfootprint*.

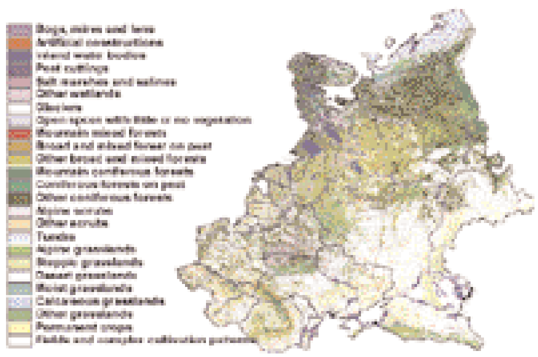


Figure 4. Habitat map of the Pan-European Ecological Network (PEEN) for Central and Eastern Europe as a basis for the PEEN indicative map. Adopted from Bouwma *et al.* 2002.

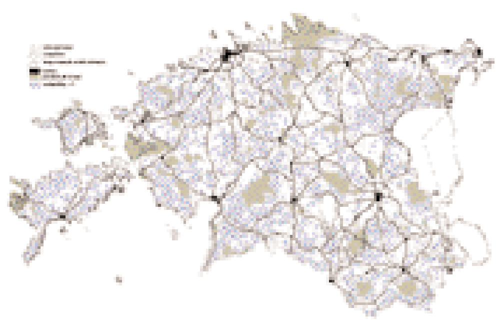


Figure 5. Suitability for the ecological network in Estonia (adopted from Remm *et al.*, 2003) as an example of an ecological network at the meso-regional (national) level. Dark grey patches indicate protected areas (relative suitability value >1.0), whereas grey areas have a suitability value of 0.5-1.0, and are mostly local core areas, various buffer zones and corridors; towns are shown in black.

zones and connected with wide corridors or stepping-stone elements (width >10 km); (c) meso-scale: small core areas (10-1000 km^2) and connecting corridors between these areas (e.g., natural river valleys, semi-natural recreation areas for local settlements; width 0.1-10 km); (d) micro-scale: small protected habitats, woodlots, wetlands, grassland patches, ponds (<10 km^2) and connecting corridors (stream banks, road verges, hedgerows, field verges, ditches; width <0.1 km; Figure 2).

The hierarchical scaling is similar to the classification of core areas based upon insights regarding the minimum required area to sustain viable populations of species (e.g., of European importance). According to this system, very large areas (critical size: >5 km^2; guarantees the long-term survival of all populations), large areas (critical size: 1-5 km^2; when isolated this area may suffer some loss of species; connection or area enlargement is required), and areas with a sub-optimal size (70-100% of species can maintain viable populations, the most demanding species can only be maintained or restored by enlargement and/or connections with comparable habitats by corridors); Bouwma *et al.*, 2002).

Mega-scale ecological networks can be considered at the global level. The Human Footprint Map can serve as a basis for determining global ecological networks (Figure 3; Sanderson *et al.*, 2002). The macro-scale of ecological networks is represented by regional-level activities like the Pan-European Ecological Network (PEEN) or national-level projects. In the Czech Republic, Slovak Republic and the Netherlands, territorial ecological networks are implemented and legislatively supported. In Estonia, Lithuania and Poland, networks are designed and some aspects accepted by law. In Hungary, Latvia, Switzerland and Ireland, network design is under development, and local or landscape-level ecological networks have been established in some parts of the territory of several European

Range of planning area	Administrative levels	Hierarchical level of core area	Diameter of core areas	Width of corridors	Planning levels in Estonia	Spatial scale (Fig. 32; Mander et al., 1995)
1–1.5*10^5 km	Earth's geographical space					
1 – 1.5*10^4 km	Geopolitical areas					
1 – 1.5*10^4 km	Group of large countries, cultural , ldistricts,large groups of countries	Global I	>1000 km	>300 km		MEGA
3 – 5*10^3 km	Large country	Global II	500 – 1000 km	200 – 300 km		MEGA
1 – 1.5*10^3 km	Group of small countries, large group of states or provinces	Regional-large	300 – 500 km	100 – 200 km		MACRO
300 – 500 km	Small country, small group of provinces or states	Regional-small	100 – 200 km	30 – 50 km	National	MACRO
100 – 150 km	Districts, small group of counties, group system of settlement groups	National-large	30 – 50 km	10 – 20 km	National District	MESO
30 – 50 km	County, large group of parishes	National-small	10 – 20 km	3 – 5 km	District	MESO
10 – 15 km	Small group of parishes, large town	District (county)-largebig	3 – 5 km	1 – 2 km	District Comprehensive	MESO
3 – 5 km	Parish, town, a part of large town, large group of villages	District (county)-small	1 – 2 km	300 – 500 m	Comprehensive	MESO
1 – 2 km	Part of town, settlement, countryside of protected area, group of villages	Local I	300 – 500 m	100 – 200 m	Detailed	MICRO
300 – 500 m	Larger group of buildings, quarter, village, field complexmassive	Local II	100 – 200 m	30 – 60 m	Detailed	MICRO
100 – 200 m	Countryside, the group of buildings with it's surrounding land, field, sectionpartition of forest	Detailed I	30 – 50 m	10 – 20 m	Detailed	MICRO
30 – 50 m	Homes and house with it's closer surroundings	Detailed II	10 – 20 m	3 – 6 m		MICRO
10 – 20 m	Apartment, a part of a house					MICRO
3 – 5 m	Space occupied by moving person, room					
1 – 2 m	Personal space of one person					

Table 1. Hierarchical levels of planning the ecological network.

countries such as Germany, Belgium, UK, Italy, Spain, Portugal, Russia, and the Ukraine (Bouwma *et al.*, 2002). Landscape-level ecological networks are designed or implemented on a wide range of spatial scales, from macro- and meso- to micro-scale projects. The most significant research on both species migration and dispersal, as well as on energy and material fluxes has been carried out at this level (see Forman, 1995; Farina, 2000). Likewise, the most detailed analysis and implementation schemes have been established at micro-scale (Figure 2).

Spatial hierarchy is closely associated with the planning levels of ecological networks. Table 1 presents a possible system of administrative levels, the range of planning areas, as well as the levels and size of core areas and connecting corridors. Experiences gained from the development of the concept of the ecological network in Estonia are presented as an example for the national-level approach. The challenge of the ecological-network approach is to integrate ecological principles, biodiversity, and landscape conservation requirements into spatial planning procedures and other land use practices.

Functions of territorial ecological networks

Ecological networks are viable because they provide mul-

tiple functions within a specific and often limited spatial area, and these functions can be planned, designed and managed to exist compatibly or synergistically (Jongman, 1995).

According to a broader concept, ecological networks (networks of ecologically compensating areas) preserve the following main ecological and socio-economical functions in landscapes (Mander *et al.*, 1988):

I. Biodiversity.
- Refuges for species (incl. genetic variability).
- Migration and dispersal tracts for biota.

II. Material and energy flows.
- Material accumulation, recycling and regeneration of resources.
- Barrier, filter and buffer for nutrient fluxes.
- Dispersal of human-induced energy.

III. Socio-economic development and cultural heritage.
- Supporting framework (e.g., recreation area) for settlements.
- Compensation and balancing of inevitable outputs of human society (e.g., supporting traditional rural development).

The relative importance of the ecological functions of the system of ecologically compensating areas depends on the spatial scale (Table 2). This varies, however, across both space and time. Based on the experience of landscape evaluation for regional and landscape planning in the countries of Central and Eastern Europe (Bastian & Schreiber, 1999), one can assume that the biodiversity support (refuge function) is more important at the macro-scale level than at the medium or micro-level. Larger natural areas with heterogeneous structure can support more species than medium- or small-size core areas (Caughley & Sinclair, 1994). On the other hand, as migration corridors and dispersal tracts, the medium-level corridors play a key role in connecting core areas of different scales. Accordingly, in the Human Footprint Map (Figure 3), for instance, areas of high value on the Human Influence Index (e.g., large areas in North America and densely populated Europe) still have remarkable high biodiversity with a list of species comparable to the period before significant anthropogenic pressure began. This is largely supported by the connectedness of natural core areas of different size. Material accumulation, the regeneration of resources, the filtering and buffering effects of material and energy fluxes need more space, and therefore their importance is greater on higher hierarchical levels (Table 2). On the other hand, the highest relative importance of all functions can be found at the meso-scale level, which integrates the national, landscape and some detail scale approaches (Table 2, Figure 2). This is one of the explanations – next to cost and complexity – of the relatively high number of studies and implementation experiences of ecological networks at the landscape level.

Global Human Footprint and Last of the Wild: ecological networks at a global level

The map of the Human Footprint, worked out by Columbia University, USA, is a global driver of conservation crises on the planet and may be considered as a base for ecological networks at the global level (Figure 3). Analysis of the Human Footprint Map indicates that 83% of the land's surface is influenced by one or more of the following factors: human population density greater than one person per square kilometer, location within 15 km of a road or major river, occupied by urban or agricultural land uses, within 2 km of a settlement or railway, and/or producing enough light to be regularly visible to a satellite at night. About 98% of the areas where it is possible to grow rice, wheat or maize (according to FAO estimates) are similarly influenced. Summarized factors have been used as

Table 2. Relative importance of the effects of ecological and socio-economic function classes of system of ecologically compensating areas at different scales.

Functions	Macro-scale	Meso-scale	Micro-scale
Biodiversity			
Refuges for species (incl. genetic variability)	high	medium	low
Migration and dispersal tracts for biota	low	high	medium
Material and energy flows			
Material accumulation, recycling and regeneration of resources	high	medium	low
Barrier, filter and buffer of nutrient fluxes	low	medium	high
Dispersal of human-induced energy	high	medium	low
Socio-economical development and cultural heritage			
Supporting framework (e.g., recreation area) for settlements	low	high	medium
Compensation and balancing of inevitable outputs of human medium society (e.g., supporting traditional rural development)	high	low	

Human Influence Index that is the quantitative base of the Human Footprint Map (Sanderson *et al.*, 2002). However, human influence is not an inevitably negative impact – for instance, the hierarchical concept of ecological networks (ecological infrastructure) shows remarkable solutions that allow people and wildlife to co-exist. Nature is often resilient if given half a chance. Hopefully, human beings will be in the position to offer or withhold that chance.

The map of the Last of the Wild, which represents the largest least influenced areas in all of the biomes of the world and in all of the world's regions (Sanderson *et al.*, 2002) is a kind of inversion of the Human Footprint map. They represent a practical starting point for long-term conservation: places where the full range of nature may still exist with a minimum of conflict with existing human structures. If we wish to conserve wildlife and wild places and have a rich and beautiful environment for ourselves, we need to find ways to diminish the negative impacts of human influence, while enhancing the positive impacts.

PEEN as an example of ecological networks at the regional level

One of the most important channels for the implementation of the Pan-European Biological and Landscapes Diversity Strategy (PEBLDS), approved by the 3rd Conference of Ministers of the Environment of 55 European countries entitled 'An Environment for Europe', held in Sofia on 25 October 1995, is the establishment of the PEEN. The participating states have agreed that the network should be established by 2005. The PEEN will contribute to achieving the main goals of the PEBLDS by ensuring that a full range of ecosystems, habitats, species and their genetic diversity, and landscapes of European importance are conserved; habitats are large enough to place species in a favourable conservation status; there are sufficient opportunities for dispersal and migration. The development programme for the PEEN will design the physical network of core areas, corridors, restoration areas and buffer zones. The programme includes the following actions: a) the elaboration of the criteria on the basis of which the network of core areas, corridors, restoration areas and buffer zones will be identified, taking the biogeographical zones of Europe into account; b) the selection of the ecosystems, habitat types, species and landscapes of European importance; c) the identification of the specific sites and corridors by way of which the respective ecosystems, habitats, species and their genetic diversity, and landscapes of European importance will be conserved and, where appropriate, enhanced or restored; d) the preparation of guidelines that will ensure that actions taken to create the network are as consistent and effective as possible. A coherent European Ecological Network of Special Areas of Conservation (SAC) is being set up under the title Natura 2000 by each of the EU Member States (as defined in the Habitats Directive (92/43/EEC Article 3). This network, composed of sites hosting the natural habitat types and species listed in Annexes I and II of the Habitats Directive, will enable the natural habitat types and the species' habitats concerned, to be maintained or, where appropriate, restored at a favourable conservation status in their natural range. However, the SAC concept considers only protected or designated areas, while the

PEEN concept also covers large undisturbed areas and their connecting corridors outside protected or designated areas. In addition, many other functions of ecological networks, such as control of energy and material fluxes, are considered by the PEEN concept.

One of the first activities of the PEEN development programme is the Indicative Map of the PEEN for Central and Eastern Europe, which is mainly based on the habitat classification and suitability analysis (Figure 4; Bouwma *et al.*, 2002).

Suitability of habitats for ecological network at national level

We consider an ecological network design to consist of three principal layers: (1) general topographical features like coastlines, the water network, major roads, and place names for locating the network portrayed, (2) habitat-based field of suitability for the ecological network, calculated from network values of landscape features using a predefined algorithm, (3) the ecological network as an administrative decision. The second layer serves as a tool supporting decision-making, while the third layer consists of the traditional components of an ecological network, such as core areas, corridors, buffer zones, and nature development/restoration areas (Remm *et al.*, 2003).

In order to create a habitat map, which served as a basis for the ecological networks suitability map, several modifications were made to the Estonian CORINE land cover map (Meiner, 1999; Remm *et al.*, 2003). All habitats, linear structures and designated areas were ranked according to their expert-assessed values (from 0 to 10) based on their naturalness, rarity and potential influence on biodiversity and landscapes. Each square on the grid (1 x 1 km) is supposed to have a certain suitability for the establishment of an ecological network (PS). The suitability of a square kilometre is determined mainly by the square's habitat structure but also by the location of the grid square relative to main migration routes of species and by management and legislation. The direction and magnitude of the influence of these factors on the PS is called the ecological network value (ENV; Remm *et al.*, 2003). We assign ENVs to the habitat classes as non-negative real numbers (e.g., 0 – presence of the factor excludes the square from the ecological network, 1 – neutral influence, 2 – twice as good as the average, the factor doubles the suitability estimation of a square 10 – the factor improves by ten times the suitability of a square). A multiplicative (logarithmic) scale is suggested because it allows the use of zero value to designate absolutely unsuitable conditions. The overall suitability [PS] of a square kilometre unit is calculated as a log product of the suitability values of all categories.

The ENV of a habitat class is given as an expert decision considering the importance of certain habitats for wildlife diversity in Estonia, and the distribution of endangered taxons in habitats according to the Red Data Book of Estonia (Remm *et al.*, 2003): The mean PS-value of a square kilometre is 0.897, and the median 1.006; the minimum value is 3.648 and the maximum 3.75. The most common network suitability is between 1.0 and 1.5. As a rule, the ecological network suitability of protected areas is higher than that of non-protected areas.

The mean natural-PS value of square kilometers that contain more than 80% protected area is 1.34, and the mean natural-PS of those square kilometers that do not include protected area is 0.819. The relative amount of protected area correlates positively with natural suitability for the ecological network. Nearly one half (47.4%) of ecologically highly valuable areas (PS >1.0) are under nature protection in Estonia. On the other hand, this means that more than one half is not protected administratively (Figure 5).

Habitat mosaic of the cultural landscape: Ecological network at landscape level

Landscape level is the most integrative among all the spatial scales of ecological networks. On the one hand, there are a great many definitions and, respectively, concepts of landscape, which makes the planning aspects very comprehensive and multifunctional. In landscape ecology, most commonly a mosaic of habitats is understood as a landscape (Forman, 1995; Farina, 2000). Due to long-term human impact and land use dynamics, European landscapes have been significantly altered. Valuable habitats in coastal and alpine areas, especially various grasslands and forests, but also wetland ecosystems in Europe as a whole have decreased dramatically in area. In large territories of high-level economic development, most natural ecosystems have been destroyed and pushed to the margins by dominant land uses such as agriculture, industrial forestry and urban development. In Europe as a whole, both homogenisation and fragmentation are the main driving factors of landscape change. As a result of fragmentation, mainly relatively small and often isolated natural areas have survived. In this mosaic, and some larger and less disturbed (semi)natural ecosystems (ecologically compensating areas) and hedgerows and riparian zones connecting them create an ecological network (infrastructure) in the cultural landscape (Figure 6), supporting the multifunctional character of the landscape.

Also, marginalisation, now dominating in Eastern, Central and Northern Europe as a main driving force of landscape change, initiates the dramatic loss of valuable seminatural ecosystems (Mander & Jongman, 1998). Some of the main functional aspects of these landscapes are connectivity and connectedness (Baudry & Merriam, 1988). The former measures the species' migration and dispersal processes by which sub-populations of organisms are interconnected into a functional demographic unit: metapopulations on different equilibrium levels (Hanski *et al.*, 1995). Connectedness refers to the structural links between elements of the spatial structure of a landscape and can be described from mappable elements (Bouwma *et al.*, 2002). The importance of metapopulation principles, partly derived from the island biogeography theory (MacArthur & Wilson, 1967; re-published in 2001; Opdam, 1991), is the acknowledgement that the survival of species involves more than solely maintaining nature reserves; ecological linkages are needed and must be included in spatial plans. Likewise, corridors between core areas and buffers around sensitive areas can provide important control of energy/material fluxes.

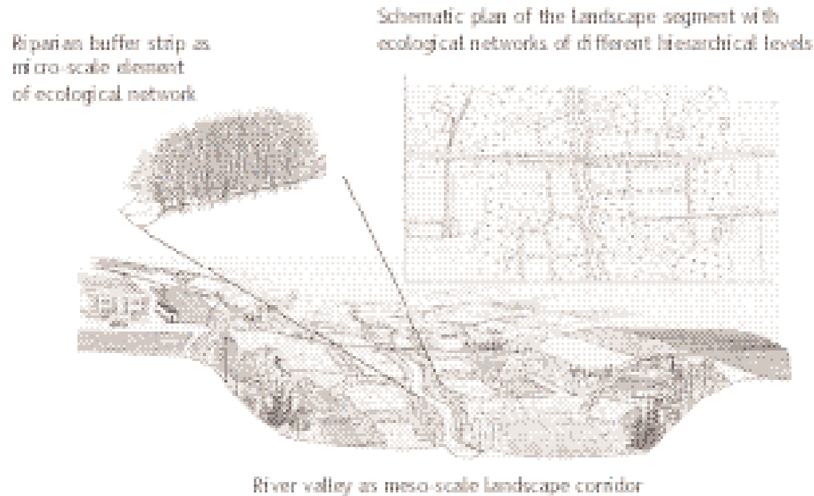


Figure 6. River valley with small-grain landscape pattern within intensively-used large-grain agricultural fields as a multifunctional landscape corridor. Hedgerows and other ecologically compensating areas in the traditional agricultural landscape of the river valley serve as examples of the ecological network at the micro-scale.

Riparian buffer zones as ecological network at micro-level

Riparian buffer zones are often considered to be multifunctional elements of rural landscapes that serve as examples of ecological networks at the most detailed level. In agricultural areas of Estonia, the preferable land-use alternative is perennial grassland (buffer zone) in combina-

tion with a forest or bush buffer strip directly on river banks or lake shores (Mander *et al.*, 1997). In some countries the complex structure of buffer zones is officially recommended or legislatively stated. For instance, in the U.S., the recommended complex buffer zone consists of three parts which are perpendicular to the stream bank or lake shore (sequentially from agricultural field to water body): a grass strip, a young (managed) forest strip and an old (unmanaged) forest strip (Lowrance *et al.*, 1984). Riparian buffer zones have the following essential functions: (1) filtering of polluted overland and subsurface flow from intensively managed adjacent agricultural fields; (2) protecting the banks of water bodies against erosion; (3) filtering polluted air, especially from local sources (*e.g.*, large farm complexes, agrochemically treated fields); (4) avoiding intensive growth of aquatic macrophytes by canopy shading; (5) improving the microclimate in adjacent fields; (6) creating new habitats in land/inland water ecotones; and (7) creating greater connectivity in landscapes due to migration corridors and stepping-stones (Mander *et al.*, 1997).

According to the hierarchy level of ecological networks, the relevance of buffer functions differs significantly. For instance, the impact of the shading effect is extremely local. Likewise, water and bank protection functions are very important on the micro-scale (local level of one or a small group of fields) and have no significant relevance on a regional, i.e. macro-scale. On the other hand, biological functions like creation of connectivity in landscapes due to migration corridors and stepping-stones is more relevant on higher hierarchical levels (Mander, 2001).

Filtering of polluted overland and subsurface flow is the key function of buffer zones (Peterjohn & Correll, 1984; Pinay & Décamps, 1988; Jordan *et al.*, 1992; Vought *et al.*, 1994). For instance, three biological processes can remove nitrogen: (1) uptake and storage in vegetation; (2) microbial immobilization and storage in the soil as organic nitrogen; and (3) microbial conversion to gaseous forms of nitrogen (denitrification: see Pinay *et al.*, 1993; Weller *et al.*, 1994; nitrification: see Watts & Seitzinger, 2000; Wolf & Russow, 2001). Various biophysical conditions control the intensity of these processes, and therefore the variability of that intensity is very high. For instance, gaseous emissions and plant uptake can vary from <1 to 1600 and from <10 to 350 kg N ha^{-1} yr^{-1}, respectively (Mander *et al.*, 1997). Thus different processes can play a leading role in nitrogen removal. The efficiency of removal also depends on input fluxes and nitrogen pools in the systems. Therefore a comprehensive budget analysis is needed to model and control the N flows in riparian ecosystems. In Figure 7, the nitrogen budget in a riparian grey alder stand is presented as an example of such modeling (Mander *et al.*, 2003).

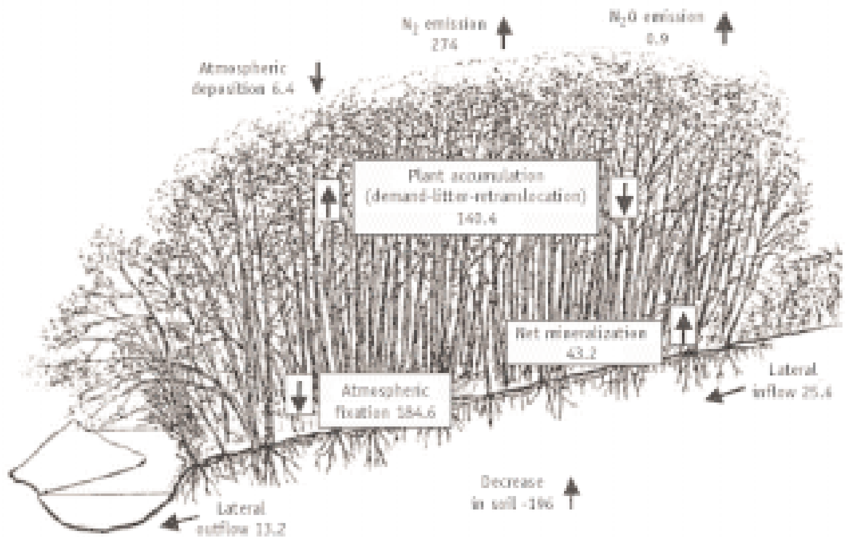


Figure 7. Nitrogen budget of a 15-year riparian grey alder stand (kg ha^{-1} yr^{-1}) as an example of the buffering function of ecological network elements (corridors and buffers) at the micro-scale level. Adopted from Mander *et al.*, 2003.

Discussion and conclusions

Emerging from the examples of ecological networks at different hierarchical levels, the following common principles can be highlighted. First, the most important and specific principle of ecological networks is *connectivity*. Together with connectedness, these are the main functional aspects in the landscape that are of importance for the dispersal and persistence of populations, and the supporting/controlling of the flow of water, the flux of nutrients, and human movement. According to Baudry and Merriam (1988) connectivity is a parameter of landscape function, which measures the processes by which sub-populations of organisms are interconnected into a functional demographic unit. Connectedness refers to the structural links between elements of the spatial structure of a landscape, which can be described from mappable elements. Sometimes biological connectivity (e.g. functional patterns) and landscape connectedness (e.g., physical connection of similar landscape elements) match, as in the movements of small forest mammals along wooded fencerows from one woodlot to another (Henein and Merriam 1990). Sometimes they do not match, as in the case of ballooning spiders (Asselin and Baudry 1989).

Structural elements differ from functional parameters. For some species connectivity is measured in the distance between sites, whereas for other species the structure of the landscape and connectedness through hedgerows represents the presence of corridors and barriers. Area reduction will cause a reduction of the populations that can survive, and in this way an increased risk of extinction. It also will increase the need for species to disperse between sites through a more or less hostile landscape.

Second, the principle of *multifunctionality* states that ecological networks always bear several functions, which are coherent to landscape functions at the relevant hierarchical level (see Bastian & Schreiber, 1999). Therefore the planning of networks following only one principle (dispersal and migration of species) may mislead the planning purposes.

Third, the principle of *continuity* means that the functioning of a network at a certain hierarchical level is only guaranteed if the full spectrum of a networks' hierarchy is performed.

In practical terms this means that ecological networks should be maintained or if necessary created at all levels. We assume that the network at lower hierarchical levels supports the biodiversity and material cycle control at the adjacent higher levels. For example, it is very complicated to support endangered species at higher scales of large areas (e.g. large and homogeneous forest plantations) if the ecological infrastructure is absent at the lower levels (e.g. meso- and micro-level habitats). Considering that principle, the hierarchical levels between adjacent levels in the hierarchy may integrate functions and characteristics prevailing at neighbouring levels. Therefore, for instance, ecological and socio-economic functions have the highest relative importance in meso-scale networks (Table 2).

Fourth, according to the principle of *plenipotentiality* (considering causal relationships between levels of hierarchy, such as causal constraints and determinations of lower-level phenomena by high-level phenomena and *vice versa*), there are no specific scale-limited functions of ecological networks. The relative importance of various functions varies depending on the hierarchical level, and planning strategies should therefore follow these variations. For instance, at the global (mega-scale) level, the leading functions of the networks are to control the global balance of CO_2 and other greenhouse gases. At the micro-level, local biodiversity support and the control of nutrient fluxes are dominant.

At the global level one art of the solution of biodiversity lies in conserving the Last of the Wild -- those few places that are relatively less influenced by human beings in all ecosystems around the globe, and give the opportunity for their connectedness (Sanderson *et al.*, 2002). It allows better stewarding of natural processes across the gradient of human influence through conservation science and action. The most important part of the solution for human beings, as individuals and through institutions and governments, however, is to moderate their influence in return for a healthier relationship with the natural world. On the other hand, at the micro-level, small-scale variations of land-use patches and their ecotones may compensate the excess nutrients.

The concept of territorial ecological networks can be considered a new paradigm in nature conservation and ecosystem management. The functions of ecological networks (biodiversity support, energy and the regulation of material fluxes, cultural and socio-economic functions) and their proportions are coherent within the hierarchy of networks. Therefore different management principles and strategy are required on different hierarchical levels. Further activities in the research, design and implementation of territorial ecological networks should concentrate on the development of coherent planning and management schemes at higher hierarchical level up to the global scale. In addition, the upscaling of ecological networks' functions and their spatial distribution is one of the priorities in the further development of this new concept of nature conservation.

Acknowledgements

This study was supported by Estonian Science Foundation Grants Nos. 692, 2471, 5261, and 5247 and Target Funding Projects Nos. 0180549s98 and 0182534s03 of the Ministry of Education and Science, Estonia. We are particularly grateful to all people and organisations who have kindly allowed us to use material from printed sources. We also thank Ms. Helen Alumäe from the Institute of Geography, University of Tartu, Estonia for her valuable comments, and Mr. Alexander Harding for proofreading the final text.

Abstract

This paper draws attention to and discusses the hierarchical nature of territorial ecological networks, and in this context their structural and functional aspects are debated. The focus of the article is on implementation and is illustrated with a number of examples, including the Pan-European Ecological Network as an example of ecological networks at the regional level and the riparian buffer zones as an ecological network at the micro-level. The upscaling and downscaling of ecological networks' functions and spatial distribution are discussed.

The paper suggests that the functions of ecological networks (biodiversity support, energy and material fluxes' regulations, cultural and socio-economic functions) and their shares depend on the level of those networks in the hierarchy. Furthermore, the functions depend on and are complementary to the simultaneous existence of ecological networks at several levels. Therefore, in land-use planning and conservation practice on different hierarchy levels, different and coordinated management principles and strategies are required.

References

Ahern, J., 1995. Greenways as planning strategy. Landscape and Urban Planning 33: 131-155.

Asselin A. and Baudry J. 1989. Les aranéides dans un espace agricol en mutation. Acta Oecologica. Oecol. Appli. 10, 143-156.

Bastian, O. & K.-F.Schreiber (Eds.), 1999. Analyse und ökologische Bewertung der Landschaft. 2. Auflage, Gustav Fischer Verlag, Jena, 564 pp.

Baudry, J. & G. Merriam, 1998. Connectivity and connectedness: functional versus structural patterns in the landscapes. In: K.-F. Schreiber (Ed.), Connectivity in Landscape Ecology. Proc. 2nd Intern. Seminar of IALE. Münstersche geographische Arbeiten 29, 23-28.

Bennett, G., 1998. Guidelines for the Development of the Pan-European Ecological Network. Draft. Council of Europe, Committee of Experts for the European Ecological Network. STRA-REP (98). 6, 35 p.

Bouwma, I.M., R.H.G. Jongman & R.O. Butovsky (Eds.), 2002. Indicative Map of the Pan-European Ecological Network for Central and Eastern Europe. Technical Background Document. ECNC Technical Report Series, ECNC, Tilburg, The Netherlands/ Budapest, Hungary, 101 pp. + annexes.

Brooker, L., M. Brooker & P. Cale, 1999. Animal dispersal in fragmented habitat: measuring habitat connectivity, corridor use and dispersal mortality. Conservation Ecology 3(1): 4 (www.consecol.org/vol3/iss1/art4/index.html).

Caughley, G. and A. R. E. Sinclair. 1994. Wildlife Ecology and Management. Blackwell Scientific Publications, Boston, MA. 334 pp.

Christaller, W., 1933. Die zentralen Orte in Süddeutschland. Gustav Fischer, Jena.

Cook, E.A., 2002. Landscape structure indices for assessing urban ecological networks. Landscape and Urban Planning 58: 269-280.

Cook, E. & H.N. van Lier, 1994. Landscape planning and ecological networks: an introduction. Landscape planning and ecological networks. Developments in Landscape management and urban planning 6F, ISOMUL, eds. E. Cook & H.N. van Lier, Elsevier: Amsterdam, Lausanne, New York, Oxford, Shannon and Tokyo, pp. 1–7.

Farina, A., 2000. Principles and Methods in Landscape Ecology. Kluwer Academic Publishers, Dordrecht, 235 pp.

Forman, R.T.T., 1995. Land Mosaics. The Ecology of Landscapes and Regions. Cambridge University Press, Cambridge, 560 pp.

Hanski, I, J. Pöyry, T. Pakkala & M. Kuussaari, 1995. Multiple equilibria in metapopulation dynamics. Nature 377: 616-621.

Henein, K. and G. Merriam, 1990. The element of connectivity where corridor is variable. Landscape Ecology 4: 157-170.

Hobbs, R., 1997. Future landscape and the future of landscape ecology. Landscape and Urban Planning, **37**, pp. 1–7.

Jongman, R.H.G., 1995. Nature conservation planning in Europe: developing ecological networks. Landscape and Urban Planning 32: 169-183.

Jongman, R.H.G. & G. Pungetti (Eds.), 2003. New Paradigms in Landscape Planning: Ecological Networks and Greenways. Cambridge University Press. (In print)

Jordan, T.E., D.L. Correll & D.E. Weller, 1992. Nutrient interception by riparian forest receiving inputs from adjacent cropland. J. Environ. Qual. 22: 467-473.

Külvik, M., K. Sepp, J. Jagomägi & Ü. Mander, 2003. Ecological networks in Estonia - from classical roots to current applications. In: Ü. Mander, & M. Antrop, Multifunctional Landscapes. Vol. III. Continuity and Change. Advances in Ecological Sciences 16. WIT Press, Southampton, Boston. In press.

Lösch, A. 1954. The Economics of Location. Yale University Press, New Haven.

Lowrance, R.R., R. Todd, J. Fail, O. Hendrickson, R. Leonard & L. Asmussen, l984. Riparian forest as nutrient filters in agricultural watersheds. BioScience 34: 374-377.

MacArthur, R.H. and O.E. Wilson, 2001. The Theory of Island Biogeography. Princeton University Press, Princeton, 224 pp.

Mander, Ü., J. Jagomägi & M. Külvik, 1988. Network of compensative areas as an ecological infrastructure of territories. In: K.-F. Schreiber (Ed.), Connectivity in Landscape Ecology. Proc. 2nd Intern. Seminar of IALE. Münstersche geographische Arbeiten 29, 35-38.

Mander, Ü., H. Palang & J. Jagomägi, 1995. Ecological networks in Estonia. Impact of landscape change. Landschap 3: 27-38.

Mander, Ü., V. Kuusemets, K. Lõhmus & T. Mauring, 1997. Efficiency and dimensioning of riparian buffer zones in agricultural catchments. Ecological Engineering 8: 299-324.

Mander, Ü., 1998. Regional assessment of the PEEN criteria in the Baltic countries. In: Support for the Establishment of the Pan-European Ecological Network in Central and Eastern Europe. Council of Europe. STRA-REP (98) 26, pp. 68-92.

Mander, Ü. & R.H.G. Jongman, 1998. Human impact on rural landscapes in central and northern Europe. Landscape and Urban Planning 41(3-4), 149-154.

Mander, Ü., 2001. Riparian Buffer Zones as Elements of Ecological Networks. In: Proc. International Workshop on Efficiency of Purification Processes in Riparian Buffer Zones. Their Design and Planning in Agricultural Watersheds. Nov. 5-9, 2001, Kushiro City, Japan. National Agricultural Research Center for Hokkaido Region, pp. 160-173.

Mander, Ü., K. Lõhmus, V. Kuusemets & M. Ivask, 2003. Nitrogen and phosphorus budgets in riparian grey alder stands. In: Ü. Mander & K. Lõhmus (Eds.), Riparian Alder Forests: Their Importance as Buffer Zones and Bioenergy Sources. Kluwer Academic Publishers, Dordrecht. (In press).

Meiner, A. (Ed.), 1999. Land Cover of Estonia. Tallinn, 133 p.

Miller, W., M.C. Collins, F.R. Steiner & E. Cook, 1998. An approach for greenway suitability analysis. Landscape and Urban Planning, 42: 91-105.

Opdam, P, 1991. Metapopulation theory and habitat fragmentation: a review of holarctic breeding bird studies. Landscape Ecology 5(2): 93-106.

Peterjohn, W.T. & D.L. Correll, 1984. Nutrient dynamics in an agricultural watershed: observations on the role of a riparian forest. Ecology 65: 1466-1475.

Pinay, G. & H. Decamps, 1988. The role of riparian woods in regulating nitrogen fluxes between the alluvial aquifer and surface water : a conceptual model. Regulated Rivers: Research and Management 2: 507-516.

Pinay, G., L. Roques, & A. Fabre, 1993. Spatial and temporal patterns of denitrification in a riparian forest. Journal of Applied Ecology 30: 581-591.

Remm, K., M. Külvik, Ü. Mander & K. Sepp, 2003. Design of the Pan-European Ecological Network: A national level attempt. In: R.H.G. Jongman & G. Pungetti (Eds.), New Paradigms in Landscape Planning: Ecological Networks and Greenways. Cambridge University Press. (In print).

Rodoman, B.B., 1974. Polarization of landscape as a manage agent in protection of biosphere and recreational resources. In: Resursy, Sreda, Rasselenije, Moscow, Nauka., p. 150-162. (In Russian).

Sanderson, E.W., M. Jaiteh, M.A. Levy, K.H. Redford, A.V. Wannebo & G. Woolmer, 2002. The Human Footprint and the Last of the Wild. BioScience 52(10): 891-904.

Saunders, D.A., R.J. Hobbs & C.R. Margules, 1991. Biological consequences of ecosystem fragmentation: a review. Conservation Biology 51: 18-32.

Thünen, J.H. von, 1990. Der isolierte Staat in Beziehung auf Landwirtschaft und Nationalökonomie, Scientia-Verlag, Aalen, 450 S.

Van Buuren, M & K. Kerkstra, 1993. The framework concept and the hydrological landscape structure: a new perspective in the design of multifunctional landscapes. In: **C.C. Vos & P.O. Opdam (Eds.)** Landscape Ecology of a Stressed Environment. Chapman and Hall, London, pp. 219-243.

Viles, R.L. & D.J. Rosier, 2001. How to use roads in the creation of greenways: case studies in three New Zealand landscapes. Landscape and Urban Planning 55: 15-27.

Vuilleumier, S. & R. Prelaz-Droux, 2002. Map of ecological networks for landscape planning. Landscape and Urban Planning 58: 157-170.

Vought, L.B.-M., J. Dahl, C.L. Pedersen & J.O. Lacoursière, 1994. Nutrient retention in riparian ecotones. Ambio 23: 342-348.

Watts, S.H. & S.P. Seitzinger, 2000. Denitrification rates in organic and mineral soils from riparian sites: a comparison of N_2 flux and acetylene inhibition methods. Soil Biol. Biochem. 32: 1383-1392.

Wolf, I. & R. Russow, 2000. Different pathways of formation of N_2O, N_2 and NO in black earth soil. Soil Biol. Biochem. 32: 229-239.

Vereniging

Het Verhaal

Het verhaal kan ons, landschapsecologen, helpen bij een methodisch probleem. Wij zijn goed in het analyseren en begrijpen van complexe systemen. Maar hoe breng ik mijn inzicht over aan de tot bestuurder gekozen burger en aan burgers die over de inrichting van hun eigen omgeving willen meedenken?

De bestuurder kan zich nog laten bijstaan door zijn deskundige. Die vergelijkt voor hem de effecten van verschillende oplossingen voor een gewenste verandering in een gebied, bijvoorbeeld voor een nieuwe waterwinning, stadswijk, industriegebied of voor meer veiligheid tegen hoogwater. De deskundige vertaalt de uitkomst van de vergelijking in een +, een 0, of een - en zet de scores in een tabel. De bestuurder vertrouwt zijn deskundige en verwijst naar hem als een kritische vraag wordt gesteld door een vertegenwoordiger van het volk. Vaak loopt die discussie stroef door het spreken in termen van goed of slecht, met de stellende vorm 'het is' in plaats van de subjectieve vorm 'ik vind'. Een voorkeur is immers per definitie subjectief. Hoe moet het dan met de kennis die nodig is om te weten wat de effecten zijn van je voorkeur? Kan je van iedere stemgerechtigde in schap, raad, staten of kamer verlangen dat hij of zij over die kennis beschikt? Op die vraag is in ons staatsbestel geen ander antwoord mogelijk dan: JA. Moeten de vertegenwoordigers dan allen gediplomeerd zijn in een hele rij disciplines, als ze willen meestemmen? Nee, dat gaat niet, maar wat dan wel?

Het goede antwoord is volgens mij: Het Verhaal.

Met het verhaal bedoel ik, dat daar alle relevante kennis in zit die nodig is om de essentie te begrijpen van het complexe systeem dat landschap heet. Wat relevant is en wat de essentie is, kan de landschapsecoloog uitleggen door zijn kennis van het systeem. Zijn inzicht wordt gevoed door zowel zijn kennis van het ecologisch functioneren van het gebied, als door zijn inzicht in de invloed van het plan op die toestand. Dat hij dat weet is mooi, maar niet genoeg voor de besluitvorming. Het zou genoeg zijn, als wij ecocratische besluiten namen. Maar daarvoor heeft onze samenleving niet gekozen. Gelukkig maar, want wij zouden als eersten ontslagen worden bij een mislukt resultaat. Wij moeten dus aan de besluitnemers onze kennis meedelen in de vorm die zij begrijpen.

Daarvoor heeft onze cultuur het verhaal. Het verhaal is daarvan zelfs de bakermat. Voordat het schrift was uitgevonden werd kennis duizenden jaren lang verzameld en doorgegeven in de vorm van het verhaal. Hoe sterk die is voor de overdracht van inzicht en kennis blijkt uit de fascinatie van de intussen opgeschreven overlevering. Tal van generaties hebben voor ons uitgedokterd, hoe iets zo te vertellen dat onze menselijke geest complexe zaken kan begrijpen.

Nu de wetenschap zo ver in de ingewikkeldheid van onze omgeving is doorgedrongen dat alleen de daarin gestudeerden het bevatten, moeten wij de draad van het verhaal weer oppakken. Daarmee emancipeert de wetenschapper zich tot burger. De burger is dank zij ons hoge opleidingsniveau mondig en vraagt daar om. Naast ons specialisme zullen wij ons ook moeten bekwamen in onze oudste cultuurvaardigheid: het maken van Het Verhaal.

JACQUES DE SMIDT, voorzitter WLO