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A hierarchical approach to ecosystems and its implications for ecological land classification : with examples of ecoregionis, ecodistrics and ecoseries of the Netherlands

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A HIERARCHICAL APPROACH TO ECOSYSTEMS AND ITS IMPLICATIONS FOR ECOLOGICAL LAND CLASSIFICATION

with examples of
ecoregions,
ecodistricts
and
ecoseries
of the Netherlands

Frans Klijn

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Preface

This thesis concerns an interdisciplinary field of study, touching upon both philosophical concepts and scientific research. Consequently, it cannot be considered as the result of one soul's thought and effort. For this reason, and because many chapters have been published previously with various co-authors, I retained the plural 'we' in all active phrases.

The general concepts and ideas for this thesis ripened during discussions with Han (J.) Runhaar, Kees (C.) L.G. Groen and Maarten I. Nip, with contributions from other colleagues of the Ecosystems and Environmental Quality section of the Centre of Environmental Science (CML-RU Leiden). Maarten van 't Zelfde was of invaluable help with his data management and GIS work.
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Summary

Introduction: the core of this contribution

This thesis is about the classification and mapping of ecosystems, essential for an integrated geographical approach to environmental management which takes into account spatial differentiation. To this end, it considers man's environment as an ecosystem, follows a systems approach and attempts to purposively connect at least three systematic angles for ecosystem analysis, *viz.*:

- ecosystem *processes* (corresponding with the central themes of environmental management);
- the vertical *structure* of main ecosystem *components* (corresponding with the media of environmental management);
- the horizontal *patterns* of ecosystems (to account for spatial variability and a geographical orientation to environmental management).

In order to understand the environment's functioning as an ecosystem we must know the relevant components and processes as well as all the *linkages* between them. In this context, we define an ecosystem as a system of structurally related abiotic and biotic components that are also functionally related by physical, chemical and biological processes. Just as there exist midges and elephants, so there exist ecosystems of various size and of differing complexity, permitting recognition of *entire* ecosystems varying from a small puddle up to the earth as a whole.
This implies that, when striving to develop one model encompassing all linkages within all ecosystems at all spatial scales, there is danger of becoming lost in the maze of complexity which follows from the fact that ecosystems may consist of numerous components with innumerable relationships among them, while they can, in addition, be studied at many spatial and temporal scales.

As a way out, we argue and consistently follow a deductive approach from theory that is based largely on the recognition of a connection between two *hierarchies*, *viz.* a *ranking* of ecosystem components according to dependence (causality) and a *scaling* of entire ecosystems according to size.
The essence of our contribution is that we use this hierarchical *ranking* of ecosystem components as a guideline for selecting classification characteristics for a *scaling* of ecosystems. This permits the development of a series of classifications for nested ecosystems for a range of practically relevant spatial scale levels in such a way that the classifications are also functionally related. In this context, the ranking ensures the connection between *components* and *processes,* while the scaling ensures the connection between *components* and *patterns* and, owing to the hierarchical ranking, also between *patterns* and *processes*.

This core of this thesis is explained in **Chapter 4**, which is preceeded by introductory chapters and followed by examples of classifications, their testing and some applications.

The context of this exercise: developments in environmental management

As described in **Chapter 1**, in the Netherlands an ecosystems approach to environmental management has evolved since 1984, when the then-minister of the environment initiated it with the policy document *'More than the sum of the parts'* (Ministry of VROM, 198-4a), in an attempt to achieve more 'integration' in environmental management. Very soon, the need for a regional orientation was also recognized (Ministry of VROM *et al.*, 1985).
Since then no fully comprehensive ecosystems approach to environmental management has evolved, however, mainly because of conceptual and, concurrently, communication problems. Against this background, this thesis aims primarily at operationalizing the *ecosystem concept* in terms of classifications and maps in such a way that they constitute *practical tools* for geographical analyses in behalf of environmental management at various relevant spatial and time scales.

The purpose of ecosystem classification and mapping

The classification of ecosystems is not a goal in itself. As explained in **Chapter 2**, the most important reason for ecosystem classification and mapping is found in the need to have access to quantified information related to surface areas, especially concerning the quality of the environment: the quality as affected by human activities and the quality in terms of significance for society.
Firstly, human activities affect different ecosystems to different degrees owing to differences in susceptibility. This requires that any classification of ecosystems should be relevant for establishing how large a part of a country or region is susceptible to specified environmental hazards and/or to what extent it is changed. Secondly, not all ecosystems are equally significant to society, nor are all changes equally bad. Therefore, ecosystem classifications which should be applicable for integrated environmental management should also be suitable for a normative assessment of nature value or changes in nature value.

In this context, we recognize two main families of applications: those of a static nature resembling physical *land evaluation*, and those of a more dynamic nature associated mainly with *environmental impact assessment* (EIA).

Classification and mapping

In **Chapter 3**, we first discuss a number of concepts and approaches to classification and mapping. In this context we propose to use the neutral *ecological land unit* to indicate the spatial extension of any ecosystem irrespective of spatial scale, whereas we restrict the term *ecotope* to ecosystems of relatively limited extent, which are homogeneous with respect to operative site factors and vegetation structure.

In theory, the density of relationships within an ecosystem in comparison to the density of

relationships with the ecosystem's surroundings should constitute the criterion for delimiting the boundaries of any ecosystem. In practice, however, ecosystems are defined on the basis of apparent homogeneity in comparison to their surroundings.
Particularly when it comes to the delimitation of larger spatial units, there are strongly deviating views. In this respect, we recognize two main groups of approaches resulting from a difference in emphasis on either the *specific heterogeneity* of certain ecological land units, often addressing these as 'landscapes' or 'ecochores', or on *homogeneities at higher scale levels* by increasing the distance to the object of study as if it were a pointillistic painting which requires stepping back. In the first approach, larger units are often mere 'chorological conglomerates' of ecotopes, which do not really differ from the 'sigma-associations' of Tüxen (1978), in which sigma literally means 'the sum' of the constituents. In the second approach, in contrast, ecological land units are delimited by emphasizing the homogeneities which result from controlling factors in the topological dimension. We follow this second approach, in which there is obviously no need to use the multi-interpretable term landscape. Maps resulting from this second approach can be regarded as hierarchical, with the more detailed maps showing the internal variability of the units defined at the scale level above. This hierarchy can be understood as a *hierarchy of nested ecosystems*.
In this approach, the different classification levels are explicitly intended for different spatial scale levels, in contrast to taxonomical classifications with a 'systematic' hierarchy irrespective of spatial scale. The clustering of spatial units at a higher systematic level in the latter method of classification does not generally result in larger spatial units. In our opinion, it is more practical to develop a series of functionally related classifications for different spatial scale levels than to try to match spatial and 'systematic' hierarchies in one single, all-encompassing classification system.

At the end of **Chapter 3** we give a brief review of a number of ecological land classification schemes from different parts of the world. This leads us to conclude that there is as yet no optimum combination of comprehensive nomenclature with unambiguous use of classification characteristics in relation to different spatial scale levels. In this context, we propose a comprehensive nomenclature for a number of spatial scale levels, based largely on Canadian and United States' nomenclatures, but including the European ecoseries and ecotopes. It comprises: *ecozones, ecoprovinces, ecoregions, ecodistricts, ecosections, ecoseries, ecotopes* and *eco-elements*.
This nomenclature has the advantage of being clearly related to the subject classified, since it begins with eco-, whereas it is also related to commonly used nomenclature for areas of different size (region, district, etc.) as used in land classification all over the world. Moreover, this nomenclature can easily be translated into almost all European languages.

Our approach in a nutshell

In **Chapter 4** we describe our approach, which is both holistic and deductive, *i.e.* from a theoretical basis. It includes the following steps:

- postulating a simplified hierarchical model of an ecosystem comprising all relevant components as functionally related subsystems, in order to structure the complexi-

ty;
- defining a classification guideline for hierarchically nested ecosystems related to this model;
- classifying and mapping ecosystems at different spatial scale levels in order to obtain multi-purpose ecosystem maps, related to each other both structurally (nested) and functionally (causally related);
- zooming in on the most relevant processes and parameters for specific applications.

A hierarchy of ecosystem components

Firstly, we discuss a simple model of an ecosystem comprising the main abiotic and biotic components in a hierarchical ranking. The model generalizes the most important linkages, in our case by hierachically ranking the components according to, among other things, a *process-functional* hierarchy. It comprises, in order of declining importance: atmosphere/climate, parent material/geology, groundwater, surface water, soil, vegetation and fauna.
The hierarchical character of this series is manifold. First and foremost it expresses that the lower components are relatively dependent on those above, but it also reflects the ecosystem patterns we can observe from the global to the local scale.

Classifying spatially nested ecosystems by controlling factors

Having defined which components belong to our conception of an ecosystem, we are confronted with the problem of classifying ecosystems with regard to all the relevant components. This is of course practically impossible unless we recognize '*correlative complexes*' (*cf.* Kwakernaak, 1982), which implies that the whole can be sufficiently known by a few of its characteristics only, owing to the fact that many characteristics are highly correlated. If we use the most discriminant characteristics at a certain spatial scale level as classification characteristics, we can estimate quite a number of other ecosystem characteristics with sufficient accuracy, especially the dependent ones.

In this context, we distinguish between classification characteristics and mapping characteristics, *i.e.* between definition and recognition. *Classification characteristics* are used to define ecosystem types and may be specified in the legend of a map, whereas all characteristics that serve as an aid for practical mapping are called *mapping characteristics*.

We argue for using *controlling factors* as *classification characteristics* for a series of related ecosystem classifications at different spatial scale levels, primarily because this yields classifications which are relevant from a process-functional point of view. As the component ranking is reflected by patterns of ecosystems of different size, nested within one another, we can use the hierarchy of ecosystem components as a guideline for classification at different spatial scale levels. The guideline reads that the prime controlling factors are likely to be characteristics of the (sets of) component(s) related to a

certain spatial scale level/classification level, *e.g.* climatic and geological characteristics for large spatial units (ecoprovinces) and characteristics of soil and vegetation for small spatial units (ecotopes). Concordingly, each classification level is specific for a pre-fixed spatial scale level, and is related to the scale levels next above and next below owing to the aforementioned process-functional linkage between the components.
This deductive approach to classification for all relevant spatial scale levels is the prime new development from which the rest follows logically.

Ecoregions and ecodistricts of the Netherlands

In **Chapter 5** we explain the mapping of ecoregions and ecodistricts of the Netherlands, a project commissioned by the Ministry of Housing, Physical Planning and Environmental Management (VROM). The aim of this exercise was to either select or establish a standardized regionalization for the Netherlands' environmental management, to be used for the integration and presentation of state-of-the-environment information in a comprehensive way.
According to our guideline, we based the *classification* of ecoregions and ecodistricts on those factors that control the pattern at mapping scales of approximately 1: 5,000,000 respectively 1: 1,000,000. The '*mapping*' was based on several dozen existing maps and geographical databases. The boundaries between individual ecoregions and ecodistricts were copied from the most relevant or accurate of these maps for each specific boundary. The other maps and atlases, especially those on biotic components, were used to assess the ecological relevance of each boundary.

The ecoregion classification for the Netherlands resulted in four terrestrial types and two aquatic types. These ecoregions were then subdivided into ecodistricts, with 26 terrestrial types and 11 aquatic types being distinguished.

Evaluating the ecoregion and ecodistrict maps

In **Chapter 6** we evaluate the consistency of use of mapping characteristics for the ecoregion and ecodistrict maps, using data on geomorphology, (vertical) groundwater movements, groundwater level classes and soils from the Landscape Ecological Mapping of the Netherlands (LKN) database. It concerns data on surface areas larger than 0.25 ha stored in the LKN format of 1 km^2 gridcells.

We used canonical variate analysis (CVA) for the evaluation. For a sample area of 2,250 gridcells around Utrecht, we first determined the discriminating value of each variable or combination of variables. Next, we automatically reclassified all gridcells with the most discriminant combination of variables, as well as the number of '*misfits*', *i.e.* the cells that are classified differently.

For the distinction between ecoregions, soil was found to be the most discriminant single variable, but combination with geomorphology yielded a substantial improvement. For the subdivion of the *Lowland peat and clay region (H)* into ecodistricts, again the combina-

tion of soil and geomorphology discriminated most, but the components changed in their order of importance. The subdivision of the *Pleistocene sand and bog region (P)* could not be well established as the sample area covered too small an area of this region.
We feel justified in stating that the ecoregion and ecodistrict maps were mapped in an adequately consistent way, since we achieved fewer than 2% misfits for the ecoregions and fewer than 10% for the ecodistricts. However, the evaluation was hampered by the fact that LKN does not cover the whole set of possibly relevant abiotic components and contains only classified data, which affects the outcome. Concordingly, we consider the results of the automated reclassification not necessarily better than our original maps but rather an aid on the road to improvement.

Susceptibility assessment of ecodistricts

In **Chapter 7** we assess the susceptibility of ecodistricts for a number of environmental hazards as a first example of an application. The procedure of susceptibility assessment closely resembles the land evaluation procedure. It requires a sound analysis of the ecological processes and ecosystem characteristics involved. This implies:

- identifying the relevant processes related to an environmental hazard;
- identifying the relevant parameters in the various components that control these processes;
- quantifying the parameter values (or classes) for the legend units and establishing their individual contribution to the control of any relevant process;
- and, finally, aggregating (or integrating) the contribution of the individual parameters into a degree of susceptibility for the legend units, thus yielding a table to convert the primary legend units into relative susceptibility classes.

We illustrate the procedure by focusing on some environmental themes that involve abiotic environmental processes, *viz.* acidification, pollution with toxic substances, eutrophication and desiccation. The results are visualized in eight different susceptibility maps.

Ecoseries classification

The ecoseries classification, which we discuss in **Chapter 8**, was initially developed for Environmental Impact Assessment (EIA), more specifically for assessing the ecological impacts of the national water management policy. This nationwide application imposed a number of practical constraints, such as poor availability of geographical data, on both the development of the classification and its operationalization in a nationwide geographical database.
The classification of ecoseries was developed primarily as an interface between human interventions and/or immissions on the one hand, and vegetation response due to changing site conditions on the other. Consequently, it had to relate closely to an existing classification of ecotopes (Stevers *et al.*, 1987; Runhaar & Udo de Haes, 1994), which is based on so-called *operative site factors* that determine the species composition of a vegetation. As most environmental hazards do not affect the operative site conditions directly, but

rather indirectly, because abiotic processes counteract the effects (*e.g.* buffering) or are triggered (*e.g.* accellerated mineralization), a related ecoseries classification based on *conditioning* soil and groundwater *characteristics* was required.

The classification characteristics were defined by scrutinizing the relationship between operative and conditioning site factors, as well as the processes resulting from environmental hazards, in order to obtain full compatibility with the existing ecotope classification in the context of predictive modelling.
Having developed a preliminary classification, we compiled a nationwide ecoseries database in a 1 km^2 grid by generalizing and converting the legends of existing data on soil and groundwater from the Landscape Ecological Mapping of the Netherlands database (De Waal, 1992), which is a generalization of the 1: 50,000 soil map.
The classification and operationalization discussed in this thesis concern the second approximation. It consists of 49 soil units with 6 groundwater table classes. In practice, only 186 relevant combinations of soil units with groundwater table classes were found to exist.

The ecological significance of the ecoseries classification

Ecoseries may be supposed to have a certain predictive power as to the occurrence of ecotopes, as the classification was explicitly developed in relation to the existing ecotope classification. As explained in **Chapter 9**, this predictive power permits an interpretation of ecoseries to be made in terms of abiotic site types to be expected. These abiotic sites may support vegetations with different vegetation structures.
However, the interpretation of soil characteristics into more directly ecologically relevant operative site factors is not unambiguous because of the internal heterogeneity of mapping units and the susceptibility of operative site factors to human disturbance. Instead of relating one operative site type to a mapping unit, we specify a range of site types with their estimated frequency distribution, which is a more probabilistic approach than commonly applied.

For this interpretation we made a conversion key, which consists of so-called *site diagrams* for each ecoseries type. The site diagrams are an indispensable complement to the ecoseries classification, as they provide the link to operative site types.
The site diagrams allow a conversion of the nationwide ecoseries database into '*abiotic site maps*'. These maps can be usefully applied for nature conservation and/or 'nature development' questions.

Evaluating the ecoseries classification

In **Chapter 9** we also evaluate the ecological significance of the ecoseries classification and concomitant site diagrams by comparing the predicted site conditions with the site conditions indicated by plant species occurrence. The evaluations are based on the assumption that the plant species composition of the vegetation is an accurate indication of the present site conditions. This assumption relies on the *ecological species groups*, as

recognized in the ecotope classification (Runhaar *et al.*, 1987).
Tests were performed for both individual site diagrams and for the nationwide abiotic site maps based on these.

For the testing of the individual site diagrams, we used 825 vegetation relevés from undisturbed areas (nature reserves and forests). This allowed site diagrams to be drawn up with found frequency distributions for 9 ecoseries types. We considered the match with the estimated site diagrams sufficient.
For evaluating the abiotic site maps, we used floristic data from FLORBASE (version 0) and vegetation data from LKN to determine the geographical distribution of about 30 aquatic, woody and herbaceous vegetations of a number of abiotic site types. These were compared with the predicted distribution of the corresponding abiotic site types. The maps showed remarkably similar distribution patterns with, in all but one case, very significant overlap percentages.
The good correlations are considered to support the approach to classification and to prove the soundness and usefulness of the ecoseries classification as such.

Applying ecoseries in hydro-ecological modelling

As mentioned before, the ecoseries classification was designed primarily for application in a nationwide predictive model (DEMNAT; Witte *et al.*, 1992a) for assessing changes in the nature value of terrestrial ecosystems resulting from water management interventions. This is the subject of **Chapter 10**.
Among other things, DEMNAT was used for an environmental impact assessment for the national Policy Plan on Drinking and Industrial Water Supply. In this context, the ecological impacts of various scenarios for groundwater extraction in the next decades were analysed.

DEMNAT consists of three essential parts, namely a geographical database of the Netherlands concerning the spatial distribution of ecosystems, dose-effect functions, and a nature valuation system.
The geographical distribution of ecoseries has been operationalized from the soil database of LKN; for each square kilometre the surface area of all the ecoseries present is given in hectares. The geographical distribution of ecotopes has been determined by using the presence of species in 1 km^2 gridcells as an indication. The combination of the two classifications yields the spatial units for which all further calculations are carried out. In the geographical database, a connection has been made between the *presence* of ecotopes of a certain type and the *surface area* of ecoseries. Such a connection is needed to determine the share of the various spatial units in each gridcell for the further calculations.
Dose-effect functions have been specified for all combinations of ecotope type and ecoseries type and validated with empirical data as far as possible. The functions describe the change in species richness in relation to hydrological changes.

In DEMNAT, the classifications of both ecoseries and ecotopes are used as interfaces between the hydrological doses on the input side and the biotic response on the output

side: the ecoseries classification permits quantification of the change in operative site factors, whereas the ecotope classification permits quantification of the subsequent response of the species composition. In this sense, both the ecoseries classification and the ecotope classification are indispensable interfaces.
Additionally, the related classifications of ecoseries and ecotopes allow a proper spatial connection of ecological groups of plant species — of which we know only the occurrence in gridcells from FLORBASE, but not their precise location — to abiotic spatial units that have been properly mapped.

Reflection

For the development of ecosystem classifications for different spatial scale levels it proved especially profitable to *link* a scaling of ecological patterns, *i.e.* of nested ecosystems of various size, to a ranking of ecosystem components by their causal dependency. Thus, we could elaborate (part of) a *series of classifications* for nested ecosystems that are also *functionally* connected.

The use of *controlling factors* as classification characteristics was found to yield *practically applicable classifications*, because these ensure a good linkage with human activities — the causes of environmental change — on the one hand, and with biotic response variables on the other. The latter linkage automatically follows from the requirement that the controlling factors are directly or indirectly ecologically relevant; this is already partly ensured by the relationship to the component ranking, but requires special attention during the actual design of a classification at a certain level.

When looking back on the classifications established thus far, it appears that the *ecoseries* classification has a thorough systematic structure and proves to be a practicable typification allowing for (relatively) safe and easy application by others, whereas, in contrast, the *ecoregion and ecodistrict* classifications are much less systematic because they were designed primarily as mapping exercises in an attempt to establish standardized ecological regionalizations, and also because they are based largely on qualitative geomorphological characteristics. Firstly, this may pose problems when extending these classifications to other areas and, secondly, it is still questionable whether the functional connectedness of the classifications at the various levels can be assured; the development of an ecosection classification — intermediate between the ecodistricts and the ecoseries — is required to explore this point.

In our attempt to operationalize the ecosystem concept, we set as our primary aim to purposively connect processes (functional relationships), components (vertical structure) and patterns (horizontal structure and scale). We based this connection on the recognition of two related hierarchies, *viz.* a *ranking* of components and a connected *scaling* of nested ecosystems. We hope to have demonstrated that the recognition of these two hierarchies may contribute to the practical operationalization of the ecosystem concept for environmental management.
In this context, we would like to finish by stating that, despite their being primarily mental constructs, in our opinion hierarchies deserve serious credit because of their

structuring value; not least with respect to the classification and mapping of spatially nested ecosystems.

Samenvatting

Inleiding: de kern van deze bijdrage

Dit proefschrift gaat over ecosysteemclassificatie en -kartering, die essentieel zijn voor een geïntegreerd milieubeleid dat rekening houdt met ruimtelijke verschillen. Hiertoe wordt het milieu van de mens als ecosysteem opgevat, wordt een systeembenadering gevolgd en wordt tevens getracht tenminste drie verschillende systematische invalshoeken voor een 'ecosysteemanalyse' op een bruikbare wijze met elkaar in verband te brengen, namelijk:

- ecosysteem*processen* (overeenkomend met de centrale thema's van het milieubeleid);
- de verticale *structuur* die wordt gevormd door de belangrijkste ecosysteem*componenten* (overeenkomend met de compartimenten van het milieubeleid); en
- de horizontale *patronen* van ecosystemen (om rekening te kunnen houden met ruimtelijke variabiliteit en voor een gebiedsgerichte benadering in het milieubeleid).

Teneinde het functioneren van het milieu als ecosysteem te begrijpen moeten alle relevante componenten en processen worden gekend, alsmede alle *verbanden* daartussen. Een ecosysteem wordt in dit verband gedefiniëerd als een systeem van structureel gerelateerde abiotische en biotische componenten die ook functioneel gerelateerd zijn door fysische, chemische en biologische processen. Zoals er muggen en olifanten bestaan, bestaan er ook ecosystemen van zeer verschillende grootte en van uiteenlopende complexiteit, waardoor het mogelijk is *gehele* ecosystemen te onderscheiden variërend van een plasje water in een karrespoor tot de aarde als geheel.
Een poging één generiek model te ontwikkelen voor alle verbanden in alle mogelijke ecosystemen op alle ruimteschalen kan dan echter gemakkelijk ten onder gaan in de complexiteit die voortvloeit uit het feit dat ecosystemen kunnen bestaan uit zeer veel componenten met daar tussen ontelbaar veel relaties, terwijl ze ook nog eens op verschillende ruimte- en tijdschalen kunnen worden bestudeerd.

Als uitweg pleiten we voor een deductieve benadering vanuit theoretische concepten, die vervolgens ook consequent door ons wordt gevolgd. Het betreft een benadering die hoofdzakelijk is gebaseerd op de relatie tussen twee *hiërarchieën*, te weten een *rangorde* van componenten van ecosystemen naar afhankelijkheid (oorzakelijkheid) en een *grootte-orde* van gehele ecosystemen naar omvang.
De essentie van deze bijdrage is dat we de *rangorde* van ecosysteemcomponenten gebruiken als richtlijn voor het kiezen van classificatiekenmerken voor naar *grootte geordende* ecosystemen. Dit maakt het mogelijk een reeks classificaties op te stellen voor geneste ecosystemen op verschillende ruimtelijke schaalniveaus op zodanig wijze dat de classificaties tevens in een functioneel verband tot elkaar staan. Daarbij heeft de rangorde betrekking op het verband tussen *componenten* en *processen* terwijl de grootte-orde betrekking heeft op het verband tussen *componenten* en *patronen* en, dankzij de rangorde, ook tussen *patronen* en *processen*.

Deze kern van het proefschrift wordt uit de doeken gedaan in **hoofdstuk 4**, voorafgegaan door enkele inleidende hoofdstukken en gevolgd door voorbeelden van classificaties, hun validatie en enkele toepassingen.

De achtergrond van dit werk: ontwikkelingen in het milieubeleid

In **hoofdstuk 1** is beschreven hoe in 1984 een ecosysteembenadering haar intrede in het Nederlandse milieubeleid heeft gedaan met het verschijnen van de nota 'Meer dan de som der delen' (Ministerie VROM, 1984a), waarin werd gesteld dat een verregaande 'integratie' in het milieubeleid noodzakelijk was. Kort daarna werd ook de noodzaak van een gebiedsgerichte benadering onderkend (Ministerie VROM *et al.*, 1985).
Sedertdien is er echter geen alomvattende ecosysteembenadering voor het milieubeleid tot stand gekomen, hoofdzakelijk als gevolg van conceptuele problemen en, bijgevolg, communicatiestoornissen. Tegen deze achtergrond wordt getracht met dit proefschrift het *ecosysteemconcept* zodanig uit te werken in de vorm van classificaties en kaarten dat deze *praktische hulpmiddelen* vormen voor geografische analyses voor milieubeleid op verschillende relevante ruimte- en tijdschalen.

Het doel van ecosysteemclassificatie en -kartering

Ecosysteemclassificatie is geen doel op zich. Zoals in **hoofdstuk 2** wordt uitgelegd is de behoefte om over kwantitatieve oppervlaktegegevens te kunnen beschikken de belangrijkste reden voor ecosysteemclassificatie en -kartering. Voor het milieubeleid gaat het daarbij vooral om gegevens over de kwaliteit van het milieu in termen van haar betekenis voor de maatschappij en voorzover ze wordt beïnvloed door het handelen van de mens.
Nu is het ten eerste zo dat een zelfde menselijk handelen voor verschillende ecosystemen verschillende gevolgen heeft door verschillen in gevoeligheid tussen die ecosystemen. Dit betekent dat een ecosysteemclassificatie geschikt moet zijn om vast te stellen welk gedeelte van een land of regio gevoelig is voor milieubederf als gevolg van een bepaalde welomschreven beïnvloeding en/of in hoeverre het systeem hierdoor verandert. Ten tweede zijn niet alle ecosystemen even belangrijk voor de maatschappij en zijn niet alle veranderingen even ongewenst. Dit vereist dat ecosysteemclassificaties die zijn ontworpen voor 'geïntegreerd' milieubeleid tevens geschikt moeten zijn voor een normatieve beoordeling van natuurwaarden of veranderingen daarin.

In dit verband onderscheiden we twee families van toepassingen: statische, die als een vorm van fysische *landevaluatie* zijn te beschouwen, en dynamische, meestal in de vorm van *milieu-effectrapportage* (m.e.r.).

Classificatie en kartering

In **hoofdstuk 3** bespreken we eerst een aantal concepten en benaderingswijzen met betrekking tot classificatie en kartering. In dat verband stellen we voor het neutrale begrip *ecologische landeenheid* te gebruiken om de ruimtelijke dimensie van een ecosysteem aan

te geven onafhankelijk van de ruimteschaal. Daarentegen reserveren we het begrip *ecotoop* voor ecosystemen van relatief beperkte omvang die homogeen zijn ten aanzien van operationele standplaatsfactoren en vegetatiestructuur.

In theorie zou de dichtheid van relaties tussen de componenten binnen een ecosysteem in vergelijking tot de dichtheid van relaties tussen een ecosysteem en diens omgeving het doorslaggevend criterium moeten zijn om een ecosysteem af te grenzen. In de praktijk worden ecosystemen echter afgegrensd op grond van hun ogenschijnlijke homogeniteit in vergelijking met de omgeving.

Er bestaan vooral over de wijze van afgrenzen van ruimtelijke eenheden groter dan ecotopen sterk verschillende opinies. We onderscheiden in dit verband twee hoofdgroepen van benaderingen die voortvloeien uit een verschillende nadruk op hetzij de *specifieke heterogeniteit* van ecologische landeenheden — waarbij deze als 'landschappen' of 'ecochoren' worden aangeduid — dan wel op *homogeniteit op hogere schaalniveaus*, waarbij de afstand tot het studieobject wordt vergroot als ware het een pointillistisch schilderij dat voor een goede waarneming vereist dat een stapje achteruit wordt gedaan. In de eerste benadering zijn de grotere eenheden gemeenlijk niet meer dan 'chorologische conglomeraten' van ecotopen die niet wezenlijk verschillen van de sigma-associaties van Tüxen (1978), waarbij sigma letterlijk 'de som' van de samenstellende delen betekent. In de tweede benadering worden ecologische landeenheden daarentegen vooral afgegrensd door te letten op de homogeniteit die ontstaat door de invloed van *sturende factoren* in de topologische dimensie, zoals klimaatverschillen of grondwaterstandsverschillen. Wij volgen deze tweede benadering, waarbij een prettige bijkomstigheid is dat het gebruik van het meerduidige begrip landschap vermeden kan worden. Kaarten die op grond van deze tweede benaderingswijze worden vervaardigd kunnen als hiërarchisch worden beschouwd, waarbij de detailkaarten de interne variabiliteit van de eenheden van het niveau erboven weergeven. Deze hiërarchie kan worden beschouwd als een *hiërarchie van geneste ecosystemen*.

In deze benadering zijn de verschillende classificatieniveaus uitdrukkelijk bedoeld voor verschillende ruimtelijke schalen — en dus kaartschalen —, in afwijking van taxonomische classificaties met een zogenaamde 'systematische' hiërarchie die juist onafhankelijk van ruimteschalen zijn. Het samenvoegen van ruimtelijke eenheden op een hoger systematisch niveau volgens de laatste methode van classificatie (bijv. naaldbossen van verschillend type) leidt in het algemeen niet tot grotere ruimtelijke eenheden. Daarom is het naar onze mening praktischer om een reeks functioneel aan elkaar gerelateerde classificaties voor verschillende schaalniveaus te ontwikkelen, dan om te proberen ruimtelijke en 'systematische' hiërarchieën in één alomvattend classificatiesysteem met elkaar te verenigen.

Aan het eind van **hoofdstuk 3** geven we een beknopte beschrijving van enkele voorbeelden van ecologische landclassificatie uit verschillende delen van de wereld. Dit resulteert in het inzicht dat er nog geen optimale systematiek bestaat waarin een eenvoudige maar omvattende nomenclatuur wordt gecombineerd met een duidelijke en ondubbelzinnige richtlijn voor het gebruik van indelingskenmerken op verschillende schaalniveaus.

Tegen die achtergrond doen we een voorstel voor een nomenclatuur voor classificatie en kartering op een aaneengesloten reeks (kaart)schaalniveaus, grotendeels geïnspireerd op nomenclaturen uit Canada en de VS, maar met inbegrip van de in Europa gebruikte

termen ecoserie en ecotoop. De reeks bestaat uit: *ecozones, ecoprovincies, ecoregio's, ecodistricten, ecosecties, ecoseries, ecotopen* en *eco-elementen*.
Deze nomenclatuur heeft het voordeel dat de relatie met het object van classificatie duidelijk is, omdat alle begrippen beginnen met eco-, terwijl de uitgang van iedere term betrekking heeft op algemeen gebruikte begrippen om in grote delen van de wereld gebieden van verschillende grootte aan te geven (regio, district, etc.). Bovendien kan deze nomenclatuur gemakkelijk in vrijwel alle Europese talen worden vertaald.

Onze benadering in een notedop

In **hoofdstuk 4** beschrijven we de door ons gevolgde benadering, die tegelijk holistisch en deductief is, dat wil zeggen beginnend vanuit theorie. De benadering bestaat uit de volgende stappen:

- het postuleren van een vereenvoudigd hiërarchisch model van een ecosysteem waarin alle relevante componenten als functioneel gerelateerde subsystemen zijn opgenomen, teneinde de complexiteit vooraf te structureren;
- het formuleren van een richtlijn voor de classificatie van hiërarchisch geneste ecosystemen in relatie tot dit model;
- het classificeren en karteren van ecosystemen op verschillende ruimtelijke schaalniveaus teneinde meervoudig toepasbare ecosysteemkaarten te verkrijgen die zowel structureel (genest) als functioneel (causaal) verband met elkaar houden;
- het inzoomen op de meest relevante processen en parameters voor specifieke toepassingen.

Een hiërarchie van ecosysteemcomponenten

Eerst bespreken we een eenvoudig model van een ecosysteem bestaande uit de belangrijkste abiotische en biotische componenten in een hiërarchische rangorde. Dit model vormt een generalisatie van de belangrijkste relaties tussen componenten, in ons geval door ze hiërarchisch te ordenen volgens, onder meer, een *causaliteitshiërarchie*. Deze omvat in volgorde van afnemende belangrijkheid: atmosfeer/klimaat, moedermateriaal/geologie, landvorm/geomorfologie, grondwater, oppervlaktewater, bodem, vegetatie en fauna.
Het hiërarchisch karakter van deze reeks is meervoudig. Als belangrijkste geldt dat in het algemeen de lagere componenten relatief afhankelijk zijn van de eigenschappen van de hoger geplaatste, maar de rangorde hangt ook samen met de ruimtelijke patronen van ecosystemen die we van mondiaal tot lokaal kunnen waarnemen.

Het classificeren van ruimtelijk geneste ecosystemen met sturende factoren als indelingskenmerken

Na te hebben vastgesteld welke componenten tot onze conceptie van een ecosysteem behoren, worden we geconfronteerd met de vraag hoe ecosystemen te classificeren ten aanzien van al deze relevante componenten. Dit is slechts mogelijk door het onderkennen van het bestaan van '*correlatieve complexen*' (zie Kwakernaak, 1982), hetgeen betekent

dat het geheel voldoende kan worden gekend met behulp van slechts enkele kenmerken dankzij het feit dat vele ecosysteemkenmerken zeer sterk met elkaar gecorreleerd zijn. Indien we nu de meest onderscheidende kenmerken op een bepaald schaalniveau als indelingskenmerken gebruiken, kunnen we een groot aantal andere ecosysteemeigenschappen met een redelijke nauwkeurigheid schatten, in het bijzonder voorzover het eigenschappen van afhankelijke componenten betreft.

We maken in dit verband onderscheid tussen indelingskenmerken en karteringskenmerken, ofwel tussen definiëring en herkenning. *Indelingskenmerken* worden gebruikt om ecosysteemtypen te definiëren en kunnen worden gebruikt in de legenda bij een kaart. Daarentegen worden alle kenmerken die praktisch bruikbaar zijn voor karteringsdoeleinden aangeduid als *karteringskenmerken*.

We pleiten ervoor om *sturende factoren* op verschillende schaalniveaus als *indelingskenmerken* te gebruiken voor een reeks ecosysteemclassificaties voor verschillende kaartschalen, in hoofdzaak omdat dit classificaties oplevert die vanuit functioneel ofwel causaliteitsoogpunt relevant zijn. Juist omdat de rangorde van componenten wordt gereflecteerd in patronen van ecosystemen op verschillende schalen, kunnen we de hiërarchie van ecosysteemcomponenten gebruiken als richtlijn bij het kiezen van de meest geschikte indelingskenmerken voor ieder schaalniveau. De richtlijn luidt dat de belangrijkste sturende factoren op een bepaald schaalniveau gebruikt moeten worden voor een classificatie op het bijbehorende classificatieniveau, bijv. klimaat en moedermateriaal voor grote ruimtelijke eenheden (ecoprovincies) en bodem en vegetatie voor kleine ruimtelijke eenheden (ecotopen). Tegelijkertijd betekent dit dat ieder classificatieniveau specifiek is voor een bepaald ruimtelijk schaalniveau maar tezelfdertijd ook in verband staat tot de schaalniveaus erboven en eronder door de reeds genoemde causaliteitsrelaties tussen de componenten.
Deze deductieve en systematische aanpak van ecologische landclassificatie voor alle relevante schaalniveaus is de belangrijkste nieuwe ontwikkeling, waar de rest van dit proefschrift logisch uit voortvloeit.

Ecoregio's en ecodistricten van Nederland

In **hoofdstuk 5** behandelen we de kartering van ecoregio's en ecodistricten, zoals die is uitgevoerd in opdracht van het ministerie van Volkshuisvesting, Ruimtelijke Ordening en Milieubeheer (VROM). Het doel van de exercitie bestond eruit een gestandaardiseerde regionalisatie voor het Nederlandse milieubeleid te kiezen dan wel te ontwikkelen, vooral met het oog op eenvoudige en eenduidige integratie en presentatie van gegevens over de toestand van het milieu.
Overeenkomstig onze richtlijn hebben we de classificatie van ecoregio's en ecodistricten gebaseerd op die factoren die het patroon van ecosystemen bepalen op kaartschalen van ongeveer 1: 5.000.000 respectievelijk 1: 1.000.000. De '*kartering*' is gebaseerd op enkele tientallen bestaande kaarten en geografische gegevensbestanden. De grenzen tussen afzonderlijke ecoregio's en ecodistricten zijn daarbij overgenomen van de voor een bepaalde grens meest relevante of nauwkeurige kaart. Andere kaarten en atlassen, in het bijzonder die met betrekking tot biotische componenten, zijn daarbij geraadpleegd om de ecologische relevantie van iedere grens te kunnen vaststellen.

De ecoregioclassificatie van Nederland omvat vier terrestrische typen en twee aquatische typen. De ecoregio's zijn vervolgens onderverdeeld in ecodistricten, met 26 terrestrische typen en 11 aquatische.

Evaluatie van de ecoregio- en ecodistrictenkaart

In **hoofdstuk 6** evalueren we of het gebruik van karteringskenmerken voor de ecoregio- en ecodistrictenkaart consistent is geweest. Voor deze evaluatie hebben we gegevens gebruikt over geomorfologie, (verticale) grondwaterbewegingen, grondwatertrappen en bodems zoals die zijn opgeslagen in het databestand van de Landschapsecologische Kartering Nederland (LKN). Het betreft gegevens over oppervlakken van iedere legendaeenheid groter dan 0,25 ha, maar opgeslagen per gridcel van 1 km^2 — de voor LKN gebruikelijke resolutie.

Voor de evaluatie hebben we '*canonical variate analysis*' (CVA) gebruikt. Daarmee hebben we voor een proefgebied van 2.250 gridcellen rond Utrecht eerst vastgesteld hoe groot de verklarende waarde van iedere variabele of combinatie van variabelen was. Vervolgens is met de meest verklarende combinatie van variabelen een automatische herclassificatie van gridcellen uitgevoerd, waarna het aantal '*missers*', dat wil zeggen afwijkend geclassificeerde cellen, kon worden vastgesteld.

Het verschil tussen ecoregio's bleek van de afzonderlijke variabelen het best verklaard te worden door bodemverschillen, maar in combinatie met geomorfologie werd nog een aanzienlijk beter resultaat geboekt. De onderverdeling van *Holoceen Laag Nederland (H)* in ecodistricten bleek eveneens het best door de combinatie van bodem met geomorfologie te worden verklaard, maar nu in omgekeerde volgorde van belangrijkheid. De onderverdeling van *Pleistoceen Hoog Nederland (P)* kon niet goed worden geëvalueerd omdat een te klein deel van deze ecoregio in het proefgebied viel.
We menen te mogen concluderen dat de ecoregio- en ecodistrictenkaart op een voldoende consistente wijze waren gekarteerd, hetgeen we afleiden uit het feit dat bij de ecoregiokaart minder dan 2% van de cellen bij automatische classificatie anders werd ingedeeld, terwijl dit voor de ecodistrictenkaart minder dan 10% was. Daar tekenen we echter bij aan dat de evaluatie enige relativering vergt, omdat het LKN-bestand niet over alle door ons bij de oorspronkelijke kartering gebruikte componenten gegevens bevat, terwijl er tevens sprake is van geclassificeerde gegevens, hetgeen de uitkomsten kan beïnvloeden. Derhalve achten we het resultaat van de automatische herclassificaties niet bij voorbaat beter dan onze oorspronkelijke kaarten, maar veeleer een hulpmiddel bij verdere verbeteringen.

Gevoeligheidsbepaling van ecodistricten

In **hoofdstuk 7** wordt als een eerste voorbeeld van toepassing de gevoeligheid van ecodistricten bepaald voor een aantal milieu(sub)thema's. De procedure van gevoeligheidsbepaling lijkt sterk op de procedure van landevaluatie. Het vereist een gedegen analyse van de processen en de daarbij betrokken ecosysteemparameters. Dit impliceert:

- het bepalen van de relevante processen die met een milieu(sub)thema samenhangen;
- het bepalen van de relevante parameters in de verschillende betrokken componenten die deze processen sturen/beïnvloeden;
- het kwantificeren van de parameterwaarden (of klassen) per legenda-eenheid en het vaststellen van de bijdrage van iedere parameter aan het geheel van procesbeinvloeding;
- en tenslotte het aggregeren (of integreren) van de bijdragen van alle afzonderlijke parameters tot één mate van gevoeligheid voor iedere legenda-eenheid, zodat een conversietabel resulteert waarmee aan iedere legenda-eenheid een gevoeligheidsklasse kan worden toegekend.

We illustreren deze procedure door enkele milieu(sub)thema's nader te bekijken, waarbij abiotische processen een belangrijke rol spelen, namelijk verzuring, verontreiniging met toxische stoffen, vermesting en verdroging. De resultaten worden gepresenteerd in de vorm van een achttal verschillende gevoeligheidskaarten.

Een classificatie van ecoseries

De ecoserieclassificatie die we in **chapiter 8** bespreken, is oorspronkelijk ontwikkeld ten behoeve van milieu-effectrapportage (m.e.r.), meer in het bijzonder voor het bepalen van de ecologische gevolgen van nationaal beleid inzake waterbeheer. De beoogde landsdekkende toepassing bracht een aantal praktische beperkingen met zich mee — zoals de beperkte beschikbaarheid van geografische gegevens — voor zowel de ontwikkeling van de classificatie als de operationalisatie in de vorm van een landsdekkend geografisch gegevensbestand.
De ecoserieclassificatie is hoofdzakelijk ontworpen als een koppeling ('interface') tussen ingrepen en immissies enerzijds en vegetatierespons als gevolg van veranderende standplaatsomstandigheden anderzijds. Bijgevolg diende de indeling nauw aan te sluiten bij een reeds bestaande ecotopenclassificatie (Stevers *et al.*, 1987; Runhaar & Udo de Haes, 1994), die is gebaseerd op zogenaamde *operationele standplaatsfactoren* die de soortsamenstelling van een vegetatie tot op grote hoogte bepalen. Aangezien echter de meeste milieuthema's die operationele standplaatsfactoren niet direct beïnvloeden, maar pas door tussenkomst van een aantal abiotische processen (bijv. zuurbuffering of versnelde mineralisatie), was het nodig een classificatie van ecoseries te ontwikkelen op basis van juist *conditionerende kenmerken* van bodem en grondwater.

De indelingskenmerken zijn gekozen op grond van een analyse van enerzijds welke conditionerende factoren de operationele standplaatsfactoren bepalen en anderzijds welke conditionerende factoren de processen beïnvloeden die samenhangen met milieuthema's, zodat de indeling bij effectvoorspelling volledig compatibel is met de ecotopenclassificatie.
Na aldus een voorlopige classificatie tot stand te hebben gebracht, hebben we een landsdekkend gegevensbestand voor 1 km^2 gridcellen gecompileerd door bestanden van de Landschapsecologische Kartering Nederland betreffende bodem en grondwater, die op hun beurt gebaseerd zijn op de 1: 50.000 bodemkaart (De Waal, 1992), verder te generalise-

ren en te converteren.
De classificatie en operationalisatie van ecoseries die in dit proefschrift worden besproken, betreffen een tweede verbeterde versie. Deze bestaat uit 49 ecologische bodemeenheden met 6 grondwaterstandsklassen. In werkelijkheid blijken slechts 186 combinaties van bodemeenheid en grondwaterstandsklasse voor te komen.

De ecologische relevantie van de ecoserieclassificatie

Van ecoseries mag worden aangenomen dat ze een zekere voorspellingskracht hebben ten aanzien van het voorkomen van ecotopen, aangezien de classificatie doelbewust is ontwikkeld in aansluiting op de bestaande ecotopenindeling. Zoals in **hoofdstuk 9** is beschreven, maakt deze voorspellingskracht het mogelijk op grond van voorkomende ecoseries een verwachting op te stellen ten aanzien van het voorkomen van abiotische standplaatstypen in operationele termen. Op de abiotische standplaatstypen kunnen vervolgens vegetaties met een verschillende verticale structuur voorkomen.
Een beoordeling van conditionerende bodemeigenschappen in termen van ecologisch directer relevante operationele standplaatstypen is echter niet onproblematisch wegens, ten eerste, de interne heterogeniteit van de meeste kaarteenheden en, ten tweede, de temporele variabiliteit van operationele standplaatsfactoren, met name als gevolg van menselijke verstoring. Daarom specificeren we per ecoserietype de waarschijnlijkheid dat verschillende abiotische standplaatstypen voorkomen, in plaats van een één op één relatie tussen ecoserietype en operationele standplaats te veronderstellen. Dit is een meer probabilistische benadering dan gebruikelijk.

Voor de ecologische interpretatie hebben we een zogenaamd *standplaatsdiagram* voor ieder ecoserietype opgesteld. Deze standplaatsdiagrammen zijn een onmisbare toevoeging aan de classificatie als zodanig, omdat ze de schakel vormen met operationele standplaatstypen. Met de standplaatsdiagrammen is het mogelijk op grond van het landsdekkend ecoseriebestand '*abiotische standplaatskaarten*' te maken. Deze kunnen als hulpmiddel dienen bij beslissingen inzake natuurbescherming en/of natuurontwikkeling.

Evaluatie van de ecoserieclassificatie

In **hoofdstuk 9** wordt ook de ecologische relevantie van de ecoserieclassificatie geëvalueerd door de met de standplaatsdiagrammen voorspelde standplaatsomstandigheden te vergelijken met de door het voorkomen van plantesoorten geïndiceerde standplaatsomstandigheden. Deze vorm van evaluatie is gebaseerd op de aanname dat de soortsamenstelling van de vegetatie een betrouwbare indicatie geeft van de actuele standplaatsomstandigheden. Deze aanname berust op een indeling van de Nederlandse flora in *ecologische soortengroepen* behorend bij de ecotopenclassificatie (Runhaar *et al.*, 1987).
We hebben zowel individuele standplaatsdiagrammen beoordeeld, als landsdekkende kaarten van het voorspelde voorkomen van abiotische standplaatstypen die op het totaal aan standplaatsdiagrammen zijn gebaseerd.

Voor een beoordeling van individuele standplaatsdiagrammen is gebruik gemaakt van 825 vegetatie-opnamen uit verhoudingsgewijs ongestoorde gebieden (natuurgebieden en bossen). Met deze gegevens was het mogelijk voor 9 ecoserietypen diagrammen met een gevonden frequentieverdeling op te stellen. De overeenkomst met de verwachte waarschijnlijkheid van voorkomen was over het algemeen bevredigend.
Voor de evaluatie van de abiotische standplaatskaarten hebben we gebruik gemaakt van gegevens uit het floristische gegevensbestand FLORBASE (versie 0) en tevens van vegetatiegegevens uit het LKN-bestand. Hiermee kon het actueel ruimtelijk voorkomen van ongeveer 30 aquatische, kruid- en bosvegetaties van een aantal verschillende abiotische standplaatstypen worden vastgesteld. De resulterende kaarten zijn vervolgens vergeleken met het voorspelde ruimtelijk voorkomen van de corresponderende abiotische standplaatsen. De beide sets kaarten vertoonden een treffende gelijkenis, hetgeen werd ondersteund door een op één na zeer significante overlap.
De resultaten van de evaluatie worden door ons beschouwd als een bewijs van de juistheid van de benadering en van de bruikbaarheid van de voorliggende ecoserieclassificatie.

Toepassing van de ecoseries in hydro-ecologische modellering

Zoals eerder gereleveerd, is de ecoserieclassificatie oorspronkelijk ontworpen voor toepassing in een landsdekkend model (DEMNAT; Witte *et al.*, 1992a) om toekomstige veranderingen te kunnen berekenen in de natuurwaarde van terrestrische ecosystemen als gevolg van ingrepen in de waterhuishouding. Deze toepassing wordt besproken in **hoofdstuk 10**.
DEMNAT is onder meer gebruikt in het kader van een milieu-effectrapportage ten behoeve van het Beleidsplan Drink- en Industriewatervoorziening. In dit verband zijn de ecologische gevolgen van verschillende scenario's voor grondwateronttrekking in de volgende decennia berekend.

DEMNAT bestaat uit drie essentiële onderdelen, namelijk een geografische database met gegevens over het voorkomen van ecosystemen van verschillende typen betreffende geheel Nederland, dosis-effectrelaties en een natuurwaarderingssysteem.
De geografische verspreiding van ecoseries is vastgesteld op basis van de bodemgegevens in het LKN-bestand; voor iedere gridcel van 1 km^2 is het oppervlak van alle voorkomende ecoseries in hectares weergegeven. De verspreiding van ecotopen is afgeleid uit het voorkomen van plantesoorten in dezelfde kilometergridcellen. De combinatie van beide classificaties levert de ruimtelijke eenheden op, waarvoor alle verdere berekeningen worden uitgevoerd. In de geografische database is aldus een verbinding gelegd tussen de *aanwezigheid* van ecotopen van een bepaald type en de *oppervlakte* van ecoseries. Zo'n verbinding is noodzakelijk om het aandeel van de verschillende ruimtelijke eenheden per gridcel te kunnen vaststellen ten behoeve van de verdere berekeningen. Dosis-effectrelaties zijn gespecificeerd in de vorm van functies voor alle combinaties van ecotooptype en ecoserietype en, voorzover mogelijk, gevalideerd met empirische gegevens. De functies beschrijven de verandering in relatieve soortenrijkdom in relatie tot lokale hydrologische veranderingen.

In DEMNAT worden de classificaties van zowel de ecoseries als de ecotopen gebruikt als

koppelingen ('interfaces') tussen de hydrologische doses aan de invoerkant en de biotische respons aan de uitvoerkant: de ecoserieclassificatie maakt het daarbij mogelijk de verandering in operationele standplaatsfactoren te kwantificeren, terwijl de ecotopenclassificatie het mogelijk maakt de daarop volgende verandering in de soortsamenstelling te bepalen. Om die reden vormen beide classificaties onmisbare schakels.
Daar komt bij dat de aan elkaar gerelateerde classificaties van ecoseries en ecotopen een verantwoorde toedeling mogelijk maken van ecologische groepen plantesoorten — waarvan we wel de presentie per kilometergridcel kennen, maar niet precies weten waar ze voorkomen — aan abiotische ruimtelijke eenheden die wel fatsoenlijk zijn gekarteerd.

Reflectie

Het bleek voor de ontwikkeling van ecosysteemclassificaties voor verschillende ruimtelijke schaalniveaus voordelig om naar grootte geordende patronen van geneste ecosystemen te *verbinden* met een rangorde van ecosysteemcomponenten gebaseerd op causale afhankelijkheid. Hierdoor werden we in staat gesteld (een deel van) een *reeks classificaties* voor ruimtelijk geneste ecosystemen te ontwikkelen, die ook *functioneel* met elkaar in verband staan.

Het gebruik van *sturende factoren* als indelingskenmerken leidt tot *praktisch bruikbare classificaties*, omdat zo een adequaat verband wordt gelegd met menselijke activiteiten — de oorzaak van milieuveranderingen — aan de ene kant, en met biotische responsvariabelen aan de andere kant. Het laatste verband volgt automatisch uit de gestelde eis dat de sturende factoren direct of indirect ecologisch relevant moeten zijn; dit wordt al deels ondervangen via de rangorde van componenten, maar bij de ontwikkeling van een classificatie op een bepaald niveau is steeds weer bijzondere aandacht vereist.

Terugblikkend op de tot nu toe uitgewerkte classificaties valt op dat de *ecoserieclassificatie* een doortimmerde systematische opbouw kent en daardoor een praktisch bruikbare indeling is. Daardoor is een verantwoord en gemakkelijk gebruik door derden mogelijk. Daarentegen zijn de classificaties van *ecoregio's en ecodistricten* veel minder systematisch van opzet, omdat ze oorspronkelijk gericht zijn geweest op het vervaardigen van 'gestandaardiseerde ecologische regionalisaties' en omdat ze tevens grotendeels zijn gebaseerd op hoofdzakelijk kwalitatieve geomorfologische kenmerken. Dit kan tot problemen leiden bij uitbreiding van deze indelingen tot grotere gebieden. Ten tweede blijft het de vraag of de functionele gerelateerdheid van de classificaties op de diverse indelingsniveaus wel voldoende is, zolang nog geen volledige verbintenis van alle aan elkaar grenzende niveaus tot stand is gebracht. Hiertoe moet echter eerst een classificatie van ecosecties — intermediair tussen ecodistricten en ecoseries — worden uitgewerkt.

In onze poging het ecosysteemconcept te operationaliseren hebben we ons als hoofddoel gesteld ecosysteemprocessen (functionele relaties), ecosysteemcomponenten (verticale structuur) en ecosysteempatronen (patroon en schaal) op een bruikbare manier met elkaar in verband te brengen. We brachten dit verband tot stand door twee gerelateerde hiërarchieën te onderscheiden, namelijk een *rangorde* van componenten en een daarmee samenhangende *grootte-orde* van geneste ecosystemen. We hopen te hebben aangetoond

dat het onderscheiden van deze twee hiërarchieën kan bijdragen aan de praktische operationalisatie van het ecosysteemconcept voor het milieubeleid.
In dit verband willen we tot slot stellen dat hiërarchieën, ondanks dat het primair gedachtenconstructies zijn, naar onze mening serieuze waardering verdienen wegens hun structurerende functie; niet het minst bij de classificatie en kartering van ruimtelijk geneste ecosystemen.

1 Introduction

1.1 An ecosystems approach in the Netherlands' environmental management: a brief history

Since 1984 environmental management in the Netherlands has evolved steadily further in the direction of integration. Prior to that date, environmental management concentrated on individual environmental media, which was reflected in the co-existence of a number of policy programmes each covering a single medium. The last release of these policy programmes comprised a provisional Indicative Policy Programme on Soil (Ministry of VROM, 1983), an Indicative Policy Programme on Air 1985-1989 (Ministry of VROM, 1984) and an Indicative Policy Programme on Water 1985-1989 (Ministry of V&W & Ministry of VROM, 1985).

The then-minister of the environment, Winsemius, laid the keel for a steady development towards a more integrated approach to the environment in the policy document *'Meer dan de som der delen'* ('More than the sum of the parts'; Ministry of VROM, 1984a), immediately followed by the first Indicative Policy Programme on Environmental Management 1985-1989 (Ministry of VROM *et al.*, 1984). In these documents the question was addressed of how to achieve integration in environmental management, which comprised, among other things, a development along two lines: an effect-oriented line and a derived source-oriented line.
The effect-oriented line emphasizes the *quality* of the environment to sustain societal development, meanwhile shifting away from individual media towards so-called central *themes*, defined as 'disorders of ecological processes and cycles' (Ministry of VROM *et al.*, 1985), such as acidification or eutrophication. These central themes were officially launched in the first Indicative Programme on Environmental Management (Ministry of VROM *et al.*, 1984). In contrast, the source-oriented line focusses on controlling societal developments in order to preserve environmental quality.

The emphasis away from individual media towards an integration through environmental themes can perhaps best be illustrated by a citation from the summary of the Indicative Policy Programme on Environmental Management 1986-1990 ((Ministry of VROM *et al.*, 1985, page 5, translated by the author): 'For a view of the environmental conditions that should be met, insight is required in ecosystems as well as in the processes and cycles within these'. Thus, the idea of an (eco)systems approach was introduced in environmental policy-making.
Since then, the trend of an ecosystems approach has persevered, as reflected by the first National Environmental Survey (RIVM, 1989), which uses the central themes as its main ordering tool while linking them to spatial scale levels, as well as by the National Environmental Policy Plan 1990-1994 (Ministry of VROM *et al.*, 1989).

Apart from the tendency towards an integrated approach, another new direction should be mentioned. It relates to the need for a regional orientation, which was also mentioned in the second Indicative Policy Programme on Environmental Management (Ministry of VROM *et al.*, 1985). This need was recognized on the basis of experiences in water

management, where it was first noticed that quality standards should be differentiated for different 'water types' (Ministry of V&W & Ministry of VROM, 1985). With hindsight this may have been the beginning of an increased geographical orientation in the Netherlands' environmental management.

1.2 An ecosystems approach in environmental research

The aforementioned developments in the Netherlands' environmental policy can be traced back to some influential publications on the functioning of the environment from a systems point of view, the most controversial and consequently most influential of which was probably Lovelock's 'Gaia' (1979). In fact, Lovelock's publication can be regarded as a very convincing explanation of the functioning of control systems in general, illustrated by the functioning of the earth as such a control system thanks to its being covered with a relatively rapidly responding biosphere. It therefore deservedly attracted considerable attention from both policy-makers and scientists from various fields, who recognized a common language and knowledge basis to discuss their concerns about the environment.

If we ignore any intended or merely supposed organismic philosophy in his Gaia, Lovelock can be regarded as a contemporary ambassador of both a systems approach to the environment and as an integrator of 'environmental sciences' in the Anglo-Saxon sense (see Bowler, 1992). He can therefore be readily situated in the historical development of 'systems ecology', strongly promoted by the work of E.P. Odum (1971) and H.T. Odum (1983). Both the systems approach as such and the attempt to integrate the (natural) environmental sciences by taking an 'ecosystems perspective' deserve somewhat closer consideration in this introduction, as they form the roots of this thesis.

1.2.1 Systems

The idea of conceptualizing any object as 'a system' has had a great influence on transdisciplinary communication. The simple concepts of 'systems theory' (Von Bertalanffy, 1950; Boulding, 1956), with its components and processes, allows the analysis of almost any tangible object, from an airconditioning device to an ocean: a function which cannot be overestimated. Next, a systems approach allows an overview of a 'holon' or whole (Smuts, 1936; Zonneveld, 1994) to be gained, but it also allows for a reductionistic in-depth analysis of whatever subsystem or connected set of components in cause-effect chains (Udo de Haes, 1991; *cf.* also De Groot, 1991). Finally, it allows for mathematical modelling of the object of study, enabling future behaviour to be forecast.

This multi-purposeness of concepts and language explains the relative popularity in both the natural sciences, especially thanks to the promotion by Von Bertalanffy (1950; 1956) and the social sciences thanks to the promotion by, among others, Miller (1975; 1978). Consequently, the concepts connected with a systems approach presently provide by far the most widely proliferated unifying language for a vast variety of sciences, as well as for technology, often even unnoticed.

We regard the unifying concepts and the common language of the systems approach so advantageous that we argue for its further and consistent use, especially when attempting to connect or integrate various scientific disciplines, such as in landscape ecology (*cf.* Bakker *et al.*, 1981; J.A. Klijn, 1995) or by the environmental sciences.

1.2.2 Integrating the environmental sciences

One of the prime problems of a systems approach is the fact that any tangible object can be regarded as a system, which makes the definition of the system at stake a crucial matter. As environmental management is engaged with environmental problems, an all-encompassing analysis would require taking into account both the entire physical environment and the societal system in an economic, social, cultural and normative sense. This may be a possible basis to establish one integrated 'environmental science' (*cf.* De Groot, 1991), but we would have to include so many scientific disciplines, as well as technology and philosophy, that we may end up with a relatively superficial analysis.
Even attempts to integrate only the relatively objective parts of economy and ecology in 'physical society-environment systems' (Odum, 1983; Braat, 1992) generally remain academic exercises, because they suffer from a lack of depth or are insufficiently applicable. More recent tools for analyzing economy-environment systems with practical unifying concepts such as Substance Flow Analysis (SFA; Van der Voet, 1996) or Life Cycle Assessment (LCA; Guinée, 1995) are successful for a few specific purposes only, because they do not take into account the spatial variability of the physical environment. These methods integrate either by attributing the potential effects of discharged substances to 'environmental media' only (SFA) or by attributing all the potential effects of a product's life cycle to 'environmental themes' only (LCA). Consequently, they are incompatible with the requirements of impact assessment or risk analysis with respect to taking into account spatial differences *and* different media *and* different themes.
We may, therefore, conclude that society-environment systems are probably too complex an object for one 'generalist' to grasp without narrowing his mind to one or more relevant dimensions; in other words: only at the cost of width of scope.

For effect-oriented environmental policy analyses, however, we may cut down slightly on our pretentions and attempt to integrate 'merely' the environmental sciences in the Anglo-Saxon sense (see Bowler, 1992), *i.e.* the physical environmental sciences, comprising climatology, oceanography, geology, geomorphology, hydrology, soil science, vegetation science and others.
This kind of integration has been tried earlier, and with success, *viz.* in landscape ecology in the 'continental' European sense (Zonneveld, 1979; Vink, 1980; Leser, 1991; Haber, 1994; Naveh & Lieberman, 1994), which is often described as a marriage of ecology and physical geography. In both 'mother disciplines' systems approaches have been fruitfully applied, in ecology by *e.g.* Tansley after he coined the ecosystem concept (Tansley, 1935) and Odum (1983), and in physical geography by *e.g.* Chorley & Kennedy (1971). In fact, ecology and physical geography are both already scientific fields that integrate disciplines which focus on one environmental medium only, such as climatology, hydrology, soil science, vegetation science, etc.. Their combination yields the 'integrated environmental sciences' against which scientific background this thesis is positioned.

1.3 On the contents of this thesis

This thesis is about the classification and mapping of ecosystems, essential for an integrated geographical approach to environmental policy analysis which takes into account spatial differentiation. To this end, it builds on a conception of the environment from a systems perspective which attempts to purposively connect at least three systematic angles for analyzing the environment, viz.:

- ecosystem *processes* (corresponding with the central themes of environmental management);
- the vertical *structure* of main ecosystem *components* (corresponding with the media of environmental management);
- the horizontal *patterns* of ecosystems (to account for spatial variability and a geographical orientation to environmental management).

The approach aims at developing ecosystem classifications and maps as general analytical tools for environmental policy analyses in all relevant spatial and time frames. Therefore, it builds heavily on generalizations of knowledge into relatively simple models and guidelines. These generalizations are required for a consistent top-down approach, which allows due reflection on and relativization from the 'case'.
Also, we consistently apply hierarchical orderings, as we consider these essential for distinguishing between the more and less important in whatever study object on whatever spatial and time scale. We shall demonstrate that by means of a consistent hierarchical ordering we are able to reconcile the different space and time frames of natural and man-induced patterns and processes with an ecosystems approach to the environment.
At the same time, we are well aware of the fact that both the generalized conception of the environment as an ecosystem and the hierarchical structuring of multiscale phenomena are mental constructs: to quote from A.N. Whitehead: *'Nature doesn't come as clean as you can think it'* (Chorley & Kennedy, 1971; page 1). However, such mental constructs are essential for communication and to make environmental surveys more effective and less costly (J.A. Klijn, 1995).

The conception of the environment as an ecosystem draws heavily on earlier work in landscape ecology (Zonneveld, 1979; Vink, 1980; Bakker *et al.*, 1981; Haber, 1982) and its application to environmental management (especially Van der Maarel & Dauvellier, 1978; Haber, 1990). The systems approach was also inspired by Chorley & Kennedy (1971) and again Van der Maarel & Dauvellier (1978), whereas the abundant recognition of hierachies was inspired by Bakker *et al.* (1981) and Van der Maarel *et al.* (1978) and sustained by the recent attention in applied ecology (O'Neill, 1988; 1989; O'Neill *et al.*, 1986; 1989; Urban *et al.*, 1987; Dale *et al.*, 1989; Müller, 1992; J.A. Klijn, 1995).

The use of a hierarchical ecosystem model (Chapter 4) for deriving a guideline for hierarchical ecosystem classification at different spatial scales forms the core of this thesis. It is the prime new development from which the rest logically follows.
On this matter, however, we must admit that only after thorough elaboration, visualization and application of the guideline for classification, did we discover that Van der Maarel (1976) may have foreseen such a development, when he discussed the correlation between the spatial patterns of plant community boundaries and the prime controlling

environmental factors, illustrated with a figure on their supposed relation which is very similar to our Figure 4.3. This figure forms the running gadget for all examples of classifications and mappings in this thesis. Did, once, Van der Maarel's figure leave an imprint in my subconscious?

For the practical elaboration of the guideline into concrete classifications and maps, we were inspired by examples of ecological land classification and/or biophysical land classification and their applications from various parts of the world, too numerous to mention here. Initially, the most influential was an overview of land classification schemes by Bailey (1981), who charted a course of waypoints to previous and current US and Canadian approaches to ecological land classification (Rowe & Sheard, 1981; Bailey *et al.*; 1985; Omernik, 1995; among many others) and their applications (Hughes *et al.*, 1990; Hunsaker *et al.*, 1990; Clarke *et al.*, 1991).

By way of elaboration, we tried the guideline on three classification levels, viz. ecoregions, ecodistricts and ecoseries. These three levels will be discussed in this thesis. The ecotope classification level has been elaborated seperately by others (Stevers *et al.* 1987; Groen *et al.*, 1993; Runhaar & Udo de Haes, 1994), but it is frequently touched upon in Chapters 8, 9 and 10, because the ecoseries classification is closely connected with it. The intermediate level of ecosections has not yet been sufficiently elaborated.

1.4 On the text

This text has been composed by revising a number of recent publications by the author, many of which were co-authored by esteemed colleagues. Each chapter is, whenever relevant, accompanied by a reference to the publication(s) upon which it is primarily based.

Chapter two concerns the relation of societal systems to ecosystems and addresses, in particular, the significance of ecosystem classification and mapping for environmental policy analysis.

The third chapter is a brief discussion of concepts and theories in (landscape) ecology, which is the main field of science drawn from in this thesis. We also discuss the primarily science-theoretical concepts of system levels, organizational levels and scale levels. Additionally, this chapter also treats some practical approaches to ecosystem classification, and more particularly ecological land classification. It ends with a proposal for an unambiguous nomenclature for ecological land classification, which will be used throughout this thesis.

In Chapter four, a deductive approach to ecosystem classification is argued, a conceptual hierachical model of an ecosystem is postulated, and from this a guideline for classification is retrieved. It forms the core of the thesis, and is followed by chapters on classifications and mappings at various scale levels, as well as their applications.

In the fifth chapter, the practical classification and mapping of ecoregions and ecodistricts of the Netherlands are discussed. It concerns two 'ecological regionalizations' intended

primarily for communicating information on the environment to policy-makers. This information may be concerned with the present or expected state of the environment, for example, which can be assessed by adapted land evaluation procedures (see Chapter 2).

The rather descriptive Chapter five is followed by an evaluation of the ecoregion and ecodistrict maps in Chapter six, which focuses mainly on the consistency of their mapping.

Chapter 7 exemplifies a first practical application, as it deals with the assessment of ecodistrict susceptibility. It precedes a chapter on another classification example, viz. of ecoseries. The main reason for presenting this application first is text economy, because the classification of ecoseries is based partly on the analysis of 'environmental themes' in this chapter.

Chapter 8 discusses the classification of ecoseries and its practical operationalization into a nationwide gridcell database. This is an example of a classification which is intended primarily for use in environmental impact assessments.

Before going into such an application, Chapter 9 discusses the ecological relevance of the ecoseries classification exemplified by some test results. It also addresses the possibility of using ecoseries for potential site mapping, a means to gain insight into the possibilities for 'nature development'.

An application in environmental impact assessment is then treated in Chapter 10, which discusses the use of the classifications of both ecoseries and ecotopes (Runhaar & Udo de Haes, 1994) in a model designed to forecast the impacts of national water management measures on the vegetation.

The eleventh chapter is a reflection on the entire exercise.

2 Ecosystem classification for environmental management[1]

2.1 Ecosystems and societal systems

Environmental management requires information of various kinds. With regard to the state of the environment this information relates to the detection, the prediction and the monitoring of changes. In this context, the environment is considered as the entire physical environment, consisting of abiotic and biotic components in mutual relationship, or in other words: as an ecosystem.

Now we could regard man as a biotic component too. This implies that he is part of the ecosystem. However, man is also the cause of environmental problems, by definition, he has the ability to foresee the consequences of his actions, and we may expect him to solve them. This places man in a special position, besides his being just a member of the fauna. In addition, ecosystems are the object of study of the natural sciences. This means that by regarding man as part of the ecosystem, we would not cover all his aspects. We would not include his psychological, social and cultural sides, which may be very important for understanding his motives and actions. For these reasons, it is enlightening to distinguish between two related systems: a societal system on the one hand, and an ecosystem on the other. This distinction is especially helpful for environmental policy analysis. We realize that it is an arbitrary distinction.

Environmental management is concerned mainly with changes in the environment due to human activities. If these changes involve abiotic and biotic processes, they can be understood as ecological process-response relations. Sometimes such processes are foreign to the ecosystem, such as toxification by xenobiotic substances, while in other cases the processes occur at unnaturally high rates, such as acidification by anthropogenic acids.
The response of the ecosystem can be understood as a change of its characteristics. In turn, the changed ecosystem may form a less suitable or less attractive environment for society. Hence, a reverse influence of the environment on society must also be distinguished.
In this perception, the ecosystem can be visualized as the central module in a cause-effect chain as depicted in Figure 2.1 in which the ecosystem is related to society in two ways.

In providing policy-makers with information on the state of the environment, we may follow different approaches. Either we focus on individual characteristics of abiotic and biotic components of the environment, or we follow an ecosystems approach in an attempt to take into account all the relations between these components. This thesis is about the ecosystems approach, which is gaining ground in the environmental sciences and environ-

[1]This chapter is a slightly revised version of part of Udo de Haes, H.A. & F. Klijn, 1994. Environmental policy and ecosystem classification. Pages 1-21 in F. Klijn (ed.), 1994. *Ecosystem Classification for Environmental Management*. Kluwer Academic Publishers, Dordrecht.

mental management for a number of reasons:

- a growing recognition of the connection between abiotic and biotic components among environmental policy-makers[2];
- a growing awareness of and interest in the value of ecosystems as 'wholes', both as a resource and for their own sake;
- the higher efficiency of an integrated approach, which requires less data acquisition thanks to correlations between ecosystem characteristics[3].

This chapter is predominantly concerned with the desired applications of ecosystem classifications to environmental management. In the next chapter, we shall define the object of classification more precisely and illuminate the scientific stage on which we act.

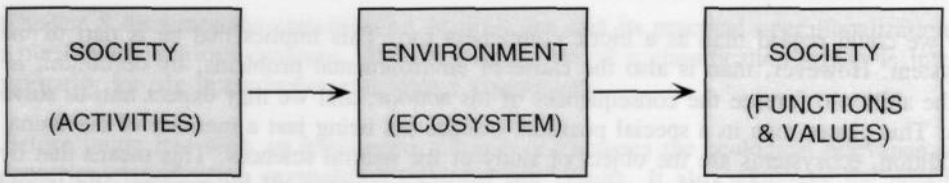

Figure 2.1 The ecosystem is related to society in two ways. Firstly, man's activities influence the ecosystem's characteristics; secondly, the changed ecosystem forms a less suitable or attractive environment for society.

2.2 On the whys of ecosystem classification and mapping

The most important reason for ecosystem classification and mapping lies in the need for access to quantified information related to surface areas. This requires mapping, which in turn requires classification. Indeed, the possibility of quantifying results in terms of surface area is the most prominent advantage of ecosystem classification and mapping over point data in monitoring.

The quantified information needed for sound environmental management relates primarily to the quality of the environment: the quality as affected by man's activities, and the quality in terms of its significance for society. To understand this, we may have a again glance at the simple effect-chain of Figure 2.1.

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[2]The disadvantages of too narrow a medium-oriented approach emerged, for example, in the definition of quality standards for individual environmental compartments, such as air, soil, and groundwater. These did not match, owing to the fact that transmittance of problems to other media was not taken into account.

[3]This is especially important in relation to field surveys and mapping. Attempts to combine information on individual components or characteristics *afterwards* often cause immense problems.

In this context, we may first recognize that man's activities affect different ecosystems to different degrees, owing to differences in the susceptibility of ecosystems to various environmental hazards. Susceptibility is a function of the abiotic and biotic processes which are disturbed or triggered. This implies that the classification of ecosystems is a relevant tool for establishing how large a part of a country or region is susceptible to environmental hazards. It requires that the classification be suitable for susceptibility assessments, and that the relevant parameters for such an assessment can be estimated or quantified.

Secondly, not all ecosystems are valued as equally significant for society, whether it be for their life-supporting functions or for their nature values. As to the life-supporting functions, these have traditionally been covered by physical land evaluation as an applied branch of land classification (FAO, 1976; Beek, 1978). The assessment of nature values is also principally a variation on land evaluation, which, however, still often struggles with the definition of goals of nature conservation and the subsequent criteria for measuring nature values (see e.g. Nip *et al.*, 1992). Generally, nature value assessments are based on biotic ecosystem components: vegetation, flora and/or fauna, because these are the most commonly appreciated. It is obvious that ecosystem classifications designed to be applicable for integrated environmental management, including nature conservation, ought to be suitable for assessing nature values.

2.3 Static and dynamic approaches to environmental policy analysis

Having discussed the relationship between the ecosystem and society in general, we may now interpret the scheme of Figure 2.1 in geographical terms. This requires that we first distinguish the spatial pattern of interventions or depositions: the loads (module 1 in Figure 2.1). Secondly, the ecosystem pattern must be known (module 2 in Figure 2.1). Finally, we must know the pattern of land use or desired land use (module 3 in Figure 2.1) which determines the requirements on ecosystem quality from, for example, land utilization and/or nature conservation points of view.

We can then draw two schemes of applications related to Figure 2.1. The first is of a '*static*' nature, especially valuable as a means to detect potential or actual environmental hazards or the deterioration of environmental quality by means of recurrent measurement. The second is of a '*dynamic*' nature and is designed for use in connection with predictive models.

In fact, the schemes of the static and the dynamic approach could be regarded as representing two families of applications: families which can be named after the traditions in which they fit best, namely the family of *physical land evaluation* on the one hand, and the family of *environmental impact assessment* (EIA) on the other (see also Hornung, 1993). Against this background, we can say that ecosystem classifications and maps for environmental management ought to be applicable in land evaluations and/or environmental impact assessments.

2.3.1 Physical land evaluation

The *static* scheme is given in Figure 2.2. In this scheme we discern three properties of ecosystems, namely susceptibility, significance and vulnerability. The *susceptibility* of an ecosystem can be defined as the degree of change resulting from a hypothetical load. It is specific for a well-defined hazard, such as acidification or groundwater lowering. In many instances, it is also necessary to specify the response variable in order to distinguish between effects on groundwater quality and topsoil quality, for example. The groundwater quality is relevant for vegetations depending on it or for public water supply, whereas a deteriorated topsoil quality may pose a risk to food chains based on either plant life or soil fauna. Susceptibility is an ecosystem property, which may be determined in an objective way. It is based on facts only, *i.e.* sheer response as measured by natural science.

The *significance* of an ecosystem is the value attributed to it by society with respect to its present and future land use function, other life-supporting functions or nature value. It is, consequently, specific for a well-defined function, such as agricultural production, fisheries, public water supply or nature conservation, and it most closely corresponds to the original concept of *suitability* (FAO, 1976; Beek, 1978). In assessing the significance of ecosystems, therefore, we are also dealing with values instead of facts only.

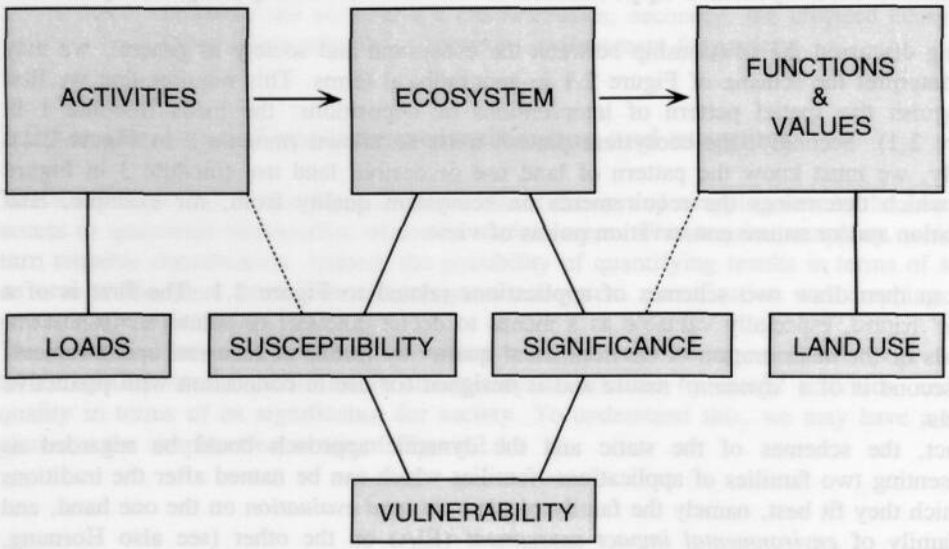

Figure 2.2 The general scheme of relations between society and ecosystem (Figure 2.1) may be interpreted in terms of relatively 'static' patterns that can be mapped. By means of *land evaluation* we may derive maps of ecosystem properties that are especially relevant for policy analysis. In each rectangle we may read 'Map of ...'.

The *vulnerability* of an ecosystem, finally, we define as the combination of susceptibility and significance (after Veelenturf, 1987). In fact, if we use the term vulnerability in connection with ecosystems, we are not only interested in the degree of change but also incorporate an answer to the question of how much we care.

The assessment of susceptibility, significance and vulnerability resembles the well-known and widely accepted procedure of physical land evaluation (FAO, 1976; Beek, 1978) already referred to in Section 2.2. It is in line with a gradual but steady broadening of the scope of land evaluation, as reflected by evolving concepts, in which suitability may be replaced by susceptibility (Várallyay *et al.*, 1989; Várallyay & Leszták, 1990; Klijn, 1991a), carrying capacity or 'environmental capacity' (De Groot, 1992), among others.
As for the procedure, suitability assessment implies the comparison of land use requirements with relevant land qualities, whereas, analogously, susceptibility assessment implies the comparison of loads with carrying capacity determinants. Whereas suitability assessment leans heavily on the definition of 'relevant and foreseeable land utilization types' (Vink, 1975), susceptibility assessment is based on relevant and foreseeable environmental hazards. Susceptibility can be determined in a relatively simple way, as described in the FAO Framework for land evaluation (FAO, 1976; see also Vink, 1975), resulting in relative classes, but it can also be assessed in a more sophisticated way, resulting in the quantification of critical loads (Nilsson, 1986; Sverdrup *et al.*, 1990; Hettelingh *et al.*, 1991; Lenz, 1994) or critical levels.

2.3.2 Environmental impact assessment

The *dynamic* scheme is shown in Figure 2.3. It is relevant in the context of predictive modelling. In this scheme, we can discern the forecasting of the effects in objective terms, the appraisal of the effects in normative terms, and the combination of both, which can be understood as comprehensive environmental impact assessment. Effect appraisal is inherently connected with impact assessment, if not by an explicit valuation step, then by the selection of relevant effect variables.

Obviously, the forecasting of effects requires *quantitative* knowledge of loads: the intensity of depositions or interventions must be known, whereas for susceptibility assessment *qualitative* knowledge suffices. Also, the pattern of depositions and interventions must be known in quantitative terms, as well as the process-response relations. Effect-forecasting therefore requires specification of both the precise type and intensity of the load and the precise type and responsiveness of the effect variable.

2.4 Summarizing the setting

This thesis is concerned with ecosystem classification for environmental management. To this end, it is clarifying to distinguish the ecosystem from a societal system. The ecosystem can then be regarded as the environment of the human society, consisting of related abiotic and biotic components. It may be negatively affected by human activities, but is also significant to society for its life-supporting functions and nature values.

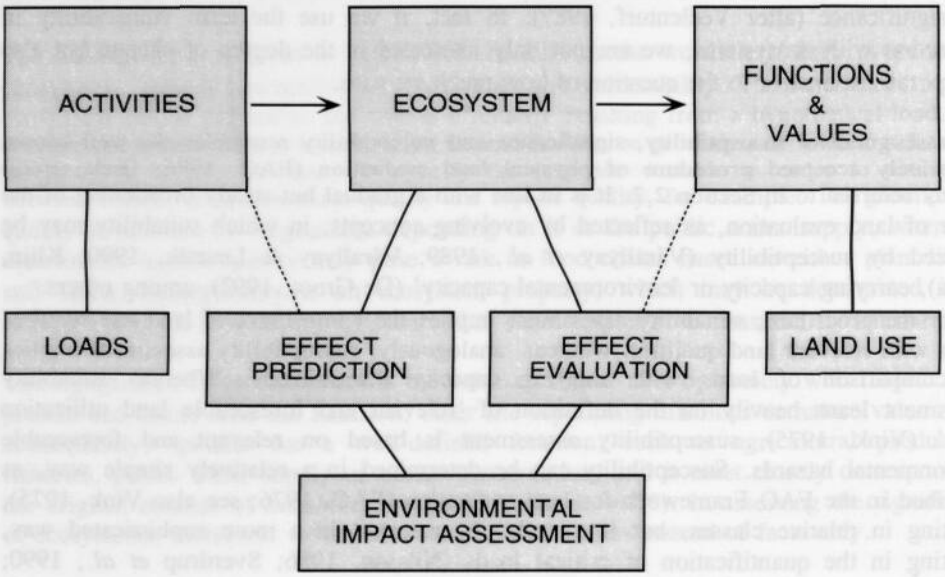

Figure 2.3 The general scheme of relations between society and ecosystem (Figure 2.1) can also be interpreted in 'dynamic' terms. In this case, the *environmental impact assessment* implies the prediction and evaluation of changes in ecosystem characteristics, which can only subsequently be expressed on maps.

The classification of ecosystems is not a goal in itself. It is mainly a tool for environmental policy analysis. In this context, we distinguish two main families of applications: those of a static nature, resembling physical land evaluation, and those of a more dynamic nature, associated mainly with environmental impact assessment. Both families of applications must meet criteria of being related to causes of ecological change on the one hand, and of allowing the appraisal of ecosystem quality by society on the other. Consequently, ecosystem classifications should preferably be related to environmental hazards ('themes'; Ministries of VROM, L&V & V&W, 1984; RIVM, 1989) and to environmental policy targets ('assessment endpoints'; Suter, 1990) at the same time.

3 The stage: concepts and practice of ecological land classification[1]

3.1 Abstract and concrete ecosystems

An ecosystem can be defined as an abstraction, *sensu* General Systems Theory (Von Bertalanffy, 1950), or as a concrete recognizable object: a tangible whole. In the abstract sense, it is a system of structurally related abiotic and biotic components, which are also functionally related by physical, chemical and biological processes (after Tansley, 1935, Chorley & Kennedy, 1971, Van der Maarel & Dauvellier, 1978, Odum, 1983). It is obvious that abstract ecosystems are not the object of this thesis. They do not exist in reality, but only in the minds of researchers and in their papers.
Here, we limit ourselves to concrete recognizable ecosystems: real-space and real-time ecosystems. Such concrete ecosystems will have to match the definition given above, but extended by criteria on how to delimit their boundaries in space and time. Delimitation in time is seldom discussed, whereas delimitation of ecosystems in space is exactly the problem we are dealing with when considering how to classify and map ecosystems.

3.2 Ecosystems in relation to system levels, organizational levels and scale levels

Having thus defined and delimited the ecosystem concept in a general way, we should first clarify its position in relation to a number of hierarchies that are frequently distinguished. In this context, three hierarchies are relevant with special reference to ecology, *viz.* those of system levels, organizational levels and scale levels.

System levels were defined during the development of General Systems Theory (Von Bertalanffy, 1950). They were adapted by Chorley & Kennedy (1971) for use in physical geography. Since system theory was developed in an attempt to achieve a unifying concept for as many sciences as possible, it is concerned mainly with the question of how to depict and understand natural wholes. This explains the levels recognized in a *hierarchy of system levels*, which run from morphological systems, via cascade systems and process-response systems to control systems with some kind of natural or artificial intelligence. As examples of the latter category, Chorley & Kennedy (1971) mention cells, individual plants and animals, ecosystems, and man-environment systems.
System levels are relevant because they relate to different explanatory mechanisms. Hence, they closely correspond to different scientific disciplines. The upper system levels always include the foregoing levels. The choice for a certain level is determined partly by

[1]This chapter is based on revised texts from Klijn, F. & H.A. Udo de Haes, 1994. A hierarchical approach to ecosystems and its implications for ecological land classification. *Landscape Ecology 9: 89-104* and Udo de Haes, H.A. & F. Klijn, 1994. Environmental policy and ecosystem classification. Pages 1-21 in Klijn, F. (ed.), 1994. Ecosystem classification for environmental management. Kluwer Academic Publishers, Dordrecht.

the object of study in the sense that each object has a highest level at which it can be studied. In contrast, we do not need to regard an object at the highest possible level. Thus, we may study a complex object in a simple way, but it must be considered senseless to regard a simple object as a complex system. On the other hand, to *fully* understand life, it is not enough to regard it as a simple cascade system. A biotic system requires an explanation by biological mechanisms. Often, however, we are not interested in full understanding, but only in partial explanations.

Organizational levels have been distinguished by, among others, Miller (1975) and Haber (1982). They are always concerned with concrete spatial and functional units: tangible wholes. The hierarchical levels are distinguished on the basis of increasing degree of organization, but the explanatory mechanisms remain largely within the same scientific field.

From Miller (1978) we could retrieve the following hierarchy of organizational levels in society: individual, group, organization, society, supranational system. These are relevant in the realm of the social sciences. Haber (1982; 1992), varying on Miller (1975), defined a hierarchy of organizational levels for the whole universe when positioning landscape ecology in the realm of ecological science. This hierarchy contained, among others, the successive levels of organism, population, community, ecosystem, landscape/ecosystem-complex, human society-environment system, ecosphere, earth ('Gaia')[2] (Haber, 1994).

There has been dispute about the sense of distinguishing the population and community levels (see Rowe, 1961; Schultz, 1967). As to population, it has been argued that this is not a 'tangible whole'. It would belong to the organizational level of the organism. With regard to communities, Tansley (1935), when he defined the ecosystem concept, already objected that these cannot be considered as wholes seperate from their abiotic environment.

These objections do not mean that populations or communities cannot be objects of study. They can, but perhaps they should not be considered as organizational levels in themselves, but as 'aspects' or subsystems of organizational levels. On the other hand, populations would fit nicely in the aforementioned hierarchy of societal organization as distinguished by Miller (1978). It is obvious that the definition of organizational levels is arbitrary and a matter of taste. We shall follow Rowe (1961) and Schultz (1967) by distinguishing only cells, organs, organisms, ecosystems and (human) society-environment systems as relevant organizational levels.

It is noticeable that in this hierarchy the ecosystem appears as a separate and individual level. According to Feibleman's *'Laws of Integrative Levels'* (Feibleman, 1954) any organizational level can be understood only by considering the levels above and below (see also Rowe, 1961; O'Neill, 1988; Müller, 1992). A certain organizational level is made up of components of the level below, whereas its significance or function can be understood only in relation to the level above. For ecosystems this would then fit nicely, because they are made up of biotic and abiotic components, while their functional significance can be understood fully only in relation to the society-environment system.

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[2]It is noticeable that there are some remarkable shifts in this list. The first shift occurs between community and ecosystem, when suddenly abiotic components are included, and the second one is caused by the appearance of man in the 'human society-environment system'.

The third hierarchy is the *hierarchy of scale levels*. These may relate to both spatial and temporal scales, which are partly connected. In a hierarchy of scale levels the size and the rate of change of the object of study differ, but no fundamentally new aspects can be discerned, which means that the system level remains the same. Also, the concrete units remain at the same organizational level.
If we apply this to ecosystems and return to the definition we gave earlier: *'systems of structurally related abiotic and biotic components that are also functionally related by physical, chemical, and biological processes'*, it is clear that ecosystems may be defined at all scales of magnitude and complexity. This implies that ecosystems can be distinguished from very small up to the earth as a whole. The organizational level and system level would remain the same, *viz.* the ecosystem as a very complex control system.
It is obvious that the distinction of scale levels is even more arbitrary than the distinction of the levels in the other two hierachies.

In practice, these three hierarchies are often confused[3]. For example, why should the ecosphere be of a different organizational level than the ecosystem? Or why should a landscape? Recently, also the discussion on the Gaia hypothesis — or should we say hypotheses? — revealed that Lovelock never intended to define *one controlling intelligence* behind his Gaia, but just pointed out that it would classify as a control system in terms of system levels[4]. In fact, we could regard Gaia as a very complex example of a control system, with many intelligences influencing its overall response, but not fundamentally different from an ecosystem. Its overall response merely resembles purposeful. This implies that, for our task of developing tools for environmental policy, we need not define a higher system level than that of a control system, nor an organizational level above the ecosystem level. It is only the scale level that differs.
In our opinion, the three hierachies should be regarded as three axes in a conceptual space. On the other hand, they are not entirely independent either, which is obvious from the fact that the control system level is relevant for biotic organizational levels such as cells or organisms, but not for the mineral composition of bedrock, which can be understood perfectly by regarding it as process-response system.

In conclusion, we repeat that ecosystems can be distinguished at many different spatial scale levels, which is important for questions of classification and mapping.

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[3]It is noticeable that Chorley and Kennedy (1971, page 4) distinguish levels of complexity *on top* of their main division of system levels, thus extending their list of system levels with self-maintaining systems, plants, animals, ecosystems, man, social systems and human ecosystems. Elsewhere, they regard these systems as *examples* of control systems. With the list on page 4 of their book they thus also add to the confusion by mixing up system levels and something like organizational levels.

[4]One of the main reasons for confusion may be the different interpretations of 'intelligence'. Control systems are defined as being controlled by some intelligence. However, this includes both artificial 'intelligence', as in a thermostat, and natural intelligence, as in living organisms. Perhaps it would help to distinguish more system levels by differentiating between *non-living* control systems, *living* control systems that are really self-maintaining and self-reproductive, and *thinking* control systems as living control systems that also act in a purposeful way. Man would classify as the latter of course, while Gaia would do perfectly as a chaotic complex of the second.

3.3 Ecosystems and mapping

The concrete ecosystems we are dealing with in this thesis have a certain spatial extension; this enables mapping. However, we should have criteria on how to distinguish them and, in second instance, how to map them.
Theoretically, the density of relations within an ecosystem as compared to the density of relations with its surroundings is the best criterion for delimiting the boundaries of an ecosystem. However, this principle is not sufficiently practicable. Common practice is much simpler. In most cases, ecosystems are defined on the basis of apparent homogeneity in comparison to their surroundings.

Thus, we can define spatial units: *ecological land units* or *ecotopes*, as they have been defined by Tansley (1939) and independently also by Troll (see Troll, 1968; 1970), who was the first to operationalize the ecotope concept.
Ecotopes are generally regarded as the smallest ecological land units that are relevant in landscape ecology. They are sometimes addressed as 'landscape cells' (Zonneveld, 1989). Ecotopes are defined as homogeneous ecological land units, the spatial expression of ecosystems predominantly determined by their structural characteristics. Van der Maarel & Dauvellier (1978) stress that ecotopes have '*a certain spatial extension*'. From the examples of ecotopes they and, for example, Neef (1967; 1970) give, it appears that ecotopes should be relatively homogeneous with respect to vegetation structure. However, above we stated that concrete ecosystems can be distinguished at many spatial scale levels: this is less strict than permitted by the requirements of 'a certain spatial extension' and homogeneity with respect to vegetation structure.
Therefore, for general use irrespective of spatial scale we advocate using the neutral *ecological land unit*, whereas the term *ecotope* should be used only for ecosystems with a homogeneous vegetation structure, *i.e.* for one spatial scale level. Runhaar and Udo de Haes (1994) discuss the ecotope concept more elaborately (see also Veen, 1982; Zonneveld, 1989).

With regard to the precise nature of ecotopes there appear no strongly deviating views. This is not the case with the delimitation of larger spatial units, however. In fact, two main groups of approaches to defining ecological land units larger than ecotopes can be discerned, resulting from a difference in emphasis on either chorological relations or topological relations. The first places emphasis on the *specific heterogeneity* of certain ecological land units, often addressing these as 'landscapes' or 'ecochores' (e.g. Haase 1976; 1989). The second emphasizes *homogeneities at larger scale levels* by increasing the distance to the object of study, as if it were a pointillistic painting which requires stepping backwards. This second group also sometimes applies the term 'landscape'.

3.3.1 Ecosystems and landscapes

For our own enterprise, we have consistently spoken of ecosystems to be classified and mapped, and not of landscapes, even though landscape ecology is sometimes defined as *'an ecosystems approach to landscapes'*. This might imply that landscapes are the 'tangible wholes' to be treated as classification objects at scale levels above that of the ecotope.

Landscapes can be defined from various points of view (see, for example, Vink, 1980; I.S. Zonneveld, 1984; J.I.S. Zonneveld, 1985): from a cultural history point of view, stressing the land use and settlement history; from a 'phsychological' point of view, emphasizing the (relation between observer and) scenery; and from an ecological point of view. These various views explain a first problem concerning the use of the term 'landscape', as the obvious existence of different definitions of landscape may cause confusion. If we limit ourselves to landscapes from an ecological point of view, however, we may again discern varying views.

If land is considered as a habitat for certain fauna species, it may be necessary to focus on relatively autonomous movements of animals between sub-habitats or functional sites (Vos *et al.* 1982). Especially animals with large home ranges often use geographical areas which need to be heterogeneous. The ecological land units that are distinguished in such a 'habitat approach' are defined on the basis of the specific heterogeneity or specific pattern characteristics which differ from those in other 'landscapes'. This requires the recognition of landscapes types and a specific approach from a faunal, *i.e.* species point of view. In this respect, landscape can be defined as a mosaic of patches (Urban *et al.* 1987) and corridors within a matrix (Forman and Godron 1986). Though this 'choric' approach *sensu* Haase (1973; 1976; 1989) belongs to landscape ecology, it is questionable whether landscape in this sense is a system, because the only functionally connecting relationship is 'the view of a species'. In our opinion it is certainly an *ecological* approach, but not a *systems* approach. Essentially, the approach is aut-ecological, which automatically implies that we do not agree with the aforementioned definition of landscape ecology being an ecosystems approach to landscapes.

Another ecological approach to landscapes is originally system-analytical (Leser, 1976; 1991; Mosimann, 1984). It clusters functionally related geosystems and biosystems again and again in order to achieve larger units in the same manner as one would constitute a personal computer from prefab components. Up to the level of ecosystem, this results in a whole, but in practice larger units often seem to be mere 'chorological conglomerates' (Zonneveld, 1984) of ecotopes. They are called landscapes but they do not really differ from the 'sigma-associations' of Tüxen (1978), where sigma literally means 'the sum' of the constituents.

In contrast, we might also delimit ecological land units by emphasizing the homogeneities which result from controlling factors in the topological dimension. The patterns on the earth surface which Walter (1979), Leser (1991, page 217) or Godron (1994) observed are entirely based on such topological relations. By focussing on these homogeneities we encounter higher ordering principles for ecological land units above ecotopes. In this specific approach, which is also system-analytical, there is, however, no need to use the term landscape, which may create confusion as to what is meant. In this thesis, we are interested in ecosystems or 'ecological land units' — as they were called in the previous section — from a systems point of view. This approach shall be examined further in this thesis, especially in the next chapter.

3.4 A hierarchical approach to ecological land classification

When classifying ecosystems, different levels of homogeneity can be distinguished for different spatial scales: from entire oceans or climatic zones to small ponds, hedgerows,

or even smaller. This is a kind of classification by subdivision (Zonneveld, 1994). For the mapping connected with the classification it means that patterns can generally be subdivided into finer patterns again and again, from global to very detailed. The corresponding maps can be regarded as hierarchical, with the more detailed maps showing the internal variability of the units defined at the scale level above. This hierarchy can be understood as a *hierarchy of nested ecosystems*, as also conceived by Bailey (1985; 1987) and comparable to the nested systems distinguished in geohydrology (Tóth, 1963; Engelen *et al.*, 1988).

For the classification and mapping of such nested ecosystems the following principles are relevant:

- when zooming in, the detail steadily increases. Classes become more narrow and the number of boundaries increases;
- boundaries that have been defined at a certain level must be retained at more detailed classification levels. The classification characteristics of higher classification levels overrule those at lower levels;
- the boundaries between already existing mapping units may, however, be defined more accurately at more detailed mapping scales.

Hierarchical classifications are by no means new. In fact, they can be encountered in the classification of soils, vegetation, geological parent material, groundwater systems, or whatever entity where the problem of spatial scale is encountered. In most cases this has resulted in the definition of hierarchical classification schemes, as will be exemplified below.

3.4.1 Spatial hierarchies versus systematic hierarchy

Sometimes, the different classification levels are explicitly intended for different spatial scale levels. In such cases we can speak of a *spatial hierarchy*. In other cases, the classification has a systematic[5] hierarchical structure, irrespective of spatial scale.
In *systematic hierarchies* all classification levels belong to one single classification. For example, in the FAO legend for soil mapping (FAO, 1988) we may find soil groups, subdivided into main and secondary soil units. In the US Soil Taxonomy (Soil Survey Staff, 1975) we find orders, sub-orders, great groups, groups, etc. Duchaufour (1977) distinguishes classes, subclasses, groups, subgroups, etc. And of course, the taxonomy of species and the taxonomy used in plant sociology are well-known examples of systematic hierarchies.
However, systematic hierarchies and spatial hierarchies do not readily match. This may be illustrated by thinking of a landscape mosaic formed by individual forest patches with different species composition among agricultural fields with different crops and grassland

[5]The use of the word 'systematic' may need some explanation. It was chosen because in our opinion it best reflects the '*systematics*' of organisms, as well as the '*syn-systematics*', which is sometimes used in vegetation science. These concepts refer to hierarchical classifications with well-defined rules of nomenclature. In this context, the term '*taxonomical*' could be used instead. Zonneveld (1994) speaks of 'a hierarchy in agglomeration' in contrast to a 'hierarchy in space'.

communities. Combining the forests with similar dominant species in order to obtain a higher level in the systematic hierarchy does not yield larger patches. Neither could, in a more natural setting, valley-bottom communities with sedges be integrated into larger units by uniting them with hillside communities with Oak or Beech forests, as distinguished in a purely systematic approach. This notion led Tüxen to define the concept of '*sigma-associations*' (Tüxen, 1978; Theurillat, 1992) with an entirely new taxonomy. In fact, this is merely an attempt to classify complexes, without changing the point of view of plant sociology.
The problem of relating systematic hierarchical classifications to spatial hierarchies often remains implicit, although Van der Maarel (1976) drew attention to the possibility of relating plant-sociological concepts to environmental spheres.

In our opinion, instead of desperately trying to match a systematic hierarchy with a spatial hierarchy, we should first look at the hierarchical approaches developed for the many integrated ecological land classifications for different spatial scale levels. In most cases, these do not consist of one single classification scheme of different systematic levels, but rather of a series of related classifications for different spatial scale levels.

3.4.2 Ecological land classification in practice

Reviewing a number of ecological land classification schemes from different parts of the world, it appears that many indeed have a hierarchical character. This holds, *inter alia*, for the Australian (Christian & Stewart, 1968), British (Brink *et al.*, 1965), Canadian (Wiken & Ironside, 1977; Wiken, 1979; Bailey *et al.*, 1985), Russian (Isachenko, 1973) and United States' (Bailey, 1976; 1981; Bailey *et al.*, 1985) classification schemes, but also for the *biomes* classification by Walter (1979) and a number of classification schemes from Central Europe (Leser, 1991; Haase, 1989). For an interesting and extensive treatment of approaches to ecological land classification in a historical perspective we refer to Bailey *et al.* (1985).

Upon closer examination, we see that some hierarchical classification systems are characterized by nicely coherent ranges of classification levels for different spatial scales such as the United States', the British and the Canadian systems, while others are quite complex in this respect. As for the classifications themselves, many ambiguities can be detected. The majority of these can be traced back to insufficient attention for the spatial scale aspects of ecosystem components. This results in the selection of classification characteristics which, in our opinion, are not suitable for mapping at the desired mapping scale. For example, the so-called Vegetation Map of Europe (Conseil de l'Europe, 1987) represents the Netherlands as being entirely covered with about three forest types. In fact, only 11 % of the Netherlands is covered by forest, and the map seems to be an interpretation of a geological map in terms of potential (climax) vegetation. In our opinion, the idea of mapping vegetation at this scale is fallacious, although we agree that the geological differences reflected on this map are ecologically relevant.

Looking at the nomenclature used for hierarchical schemes for ecological land classification, we notice a striking diversity, especially when taking into account that our review is far from complete, because no French or other Roman nomenclatures have been seriously

taken into account (*cf.* Blandin & Lamotte, 1988; Theurillat, 1992) and classification schemes from eastern European countries or Asia have not been looked at either. Still, almost all countries and even individual scientists appear to be using their own nomenclatures. Some of these have been compared by Bailey (1981), illustrated in Table 3.1. We placed a number of nomenclatures commonly used in the Netherlands in Table 3.2 in a comparable way, whereas Figure 3.1 shows the nomenclature used by Walter (1976). Still other nomenclatures can be found in Leser (1991, pages 202-203), who compares a large number of nomenclatures used in Germany including the influential proposals of Troll and Neef who distinguish a range of *chorae* from micro-chore to mega-chore.

Table 3.1 Comparison of the nomenclature of some ecological classification systems of a hierarchical character, after Bailey (1981). Comparable concepts have been placed on the same level.

AUSTRALIA	UK	CANADA	USSR	USA
			ZONE	
				DOMAIN
	LAND ZONE			DIVISION
	LAND REGION	ECOREGION	PROVINCE	PROVINCE
	LAND DISTRICT	ECODISTRICT		SECTION
			LANDSCAPE	
LAND SYSTEM	LAND SYSTEM	ECOSECTION		DISTRICT
	LAND TYPE	ECOSITE	UROCHISHCHA	LANDTYPE ASSOCIATION
LAND UNIT				
LAND TYPE	LAND PHASE			LANDTYPE
SITE		ECOELEMENT		LANDTYPE PHASE
			FACIA	SITE

The differences in nomenclature can be partly explained in terms of differences in approach, for example a more choric approach by those who study fauna relations as against a more topological approach by vegetation ecologists. Secondly, there are schools of landscape ecology issuing from, for example, various countries or universities. Thirdly, different languages may explain some of the differences. Despite these largely valid arguments for a diversity in nomenclature, we regard it as a handicap in discussions among scientists and policy-makers.

Table 3.2 Comparison of four series of nomenclature frequently used for ecological land classifications in the Netherlands (from Klijn, 1988).

1	2	3	4
	SUPERREGION		
MAIN LANDSCAPE		DISTRICT	
	SUBREGION		
		GEOTOPE	LANDSCAPE
(SUB)LANDSCAPE			
		ECOCHORE	
LAND UNIT			PHYSIOTOPE
		ECOTOPE	ECOTOPE

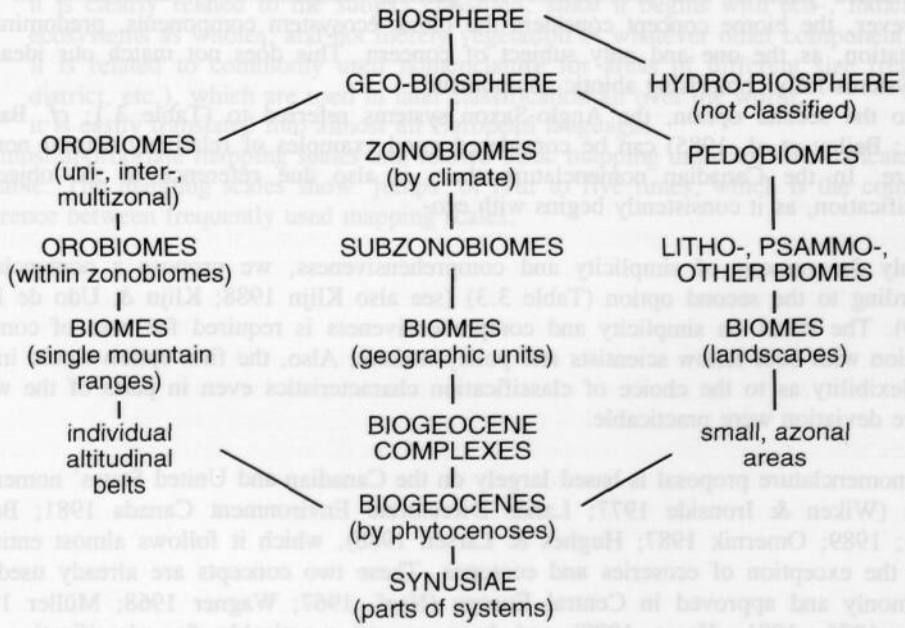

Figure 3.1 Hierarchical levels of ecosystem classification by vegetation characteristics at different spatial scales according to Walter (1979). The nomenclature reflects the determining abiotic factors in a main climatic series (centre), an orographic subseries (physiography; left) and an edaphic subseries (soil; right).

Reflecting on the above, we conclude that in most cases different nomenclatures and/or different classification characteristics have been used for different spatial scale levels. Secondly, in our opinion the optimum combination of a really comprehensive nomenclature and an unambiguous use of classification characteristics in relation to different spatial scales has not yet been found. It is one of the aims of this thesis to contribute to the search for such a classification scheme. Therefore, we first propose a comprehensive nomenclature which is connected with the conceptual framework we shall present in the next chapter.

3.5 Nomenclature proposal

As to a relatively simple and unambiguous nomenclature, there are two options: either we should specify the ecosystem component which determines the pattern of ecosystems we perceive, or we should use a relatively neutral nomenclature in combination with clear guidelines for the classification and mapping.
As to the first option, Walter (1979) may be referred to as an example. He uses the words *zono*biome for climatically determined patterns, *oro*biomes for physiographically determined patterns and *pedo*biomes for patterns determined by differences in soil. However, the biome concept considers the biotic ecosystem components, predominantly vegetation, as the one and only subject of concern. This does not match our ideas on ecosystems fully comprising abiotic components.
As to the second option, the Anglo-Saxon systems referred to (Table 3.1; *cf.* Bailey, 1981; Bailey *et al.* 1985) can be considered good examples of relatively neutral nomenclature. In the Canadian nomenclature there is also due reference to the object of classification, as it consistently begins with eco-.

Mainly for reasons of simplicity and comprehensiveness, we propose a nomenclature according to the second option (Table 3.3) (see also Klijn 1988; Klijn & Udo de Haes 1990). The stress on simplicity and comprehensiveness is required for ease of communication with both fellow scientists and policy-makers. Also, the first option would inhibit all flexibility as to the choice of classification characteristics even in parts of the world where deviation were practicable.

The nomenclature proposal is based largely on the Canadian and United States' nomenclatures (Wiken & Ironside 1977; Lands Directorate Environment Canada 1981; Bailey 1981; 1989; Omernik 1987; Hughes & Larsen 1988), which it follows almost entirely, with the exception of ecoseries and ecotopes. These two concepts are already used too commonly and approved in Central Europe (Neef, 1967; Wagner 1968; Müller 1970; Leser 1976; 1991; Haase 1989) and have proved practicable for classification and mapping. Consequently, the proposed nomenclature can be regarded as the result of selection and combination, yielding what we consider '*the best of both worlds*'.

Table 3.3 Nomenclature proposal for hierarchical ecosystem classification with indicative mapping scales and basic mapping units.

	INDICATIVE MAPPING SCALE		BASIC MAPPING UNIT		
ECOZONE	1: >	50,000,000	>	62,500	km^2
ECOPROVINCE	1: 10,000,000 -	50,000,000	2,500 -	62,500	km^2
ECOREGION	1: 2,000,000 -	10,000,000	100 -	2,500	km^2
ECODISTRICT	1: 500,000 -	2,000,000	625 -	10,000	ha
ECOSECTION	1: 100,000 -	500,000	25 -	625	ha
ECOSERIES	1: 25,000 -	100,000	1.5 -	25	ha
ECOTOPE	1: 5,000 -	25,000	0.25 -	1.5	ha
ECO-ELEMENT	1: <	5,000	<	0.25	ha

The proposed nomenclature has the following advantages:

- it is clearly related to the subject classified, since it begins with eco-, indicating ecosystems as wholes, and not merely vegetation or whatever other component;
- it is related to commonly used nomenclature for areas of different size (region, district, etc.), which are used in land classification all over the world;
- it is easily translated into almost all European languages.

The most appropriate mapping scales and related basic mapping units are also indicated in the table. The mapping scales show 'jumps' of four to five times, which is the common difference between frequently used mapping scales.

4 A hierarchical approach to ecosystems and its implications for classifying spatially nested ecosystems[1]

4.1 How to proceed in the maze of overwhelming complexity?

Ecosystems are very complex systems. They are made up of a wealth of structurally related biotic and abiotic components that are also functionally related by innumerable processes: fluxes of energy and matter. Thus, ecosystems may be studied by zooming in on different aspects. For instance, we may emphasize the structural characteristics by looking at the species composition of the vegetation in relation to controlling site factors. This is the common approach of vegetation ecologists (*cf.* Runhaar & Udo de Haes, 1994). But we may also concentrate on energy budgets (Odum, 1983), or on matter budgets (Lenz, 1994). This confronts us with the question where to begin.

For classification purposes, it seems obvious that we should look for characteristics of ecosystems that can easily be recognized or measured in the field (Zonneveld, 1979; 1994). In practice, this would mean that we could concentrate completely on structural characteristics, because processes cannot be recognized as easily.
On the other hand, the desired applications for environmental policy analyses seem to require an emphasis on processes. More specifically, on processes that result from environmental hazards. An analysis of acidification, for example, reveals that it influences numerous ecosystem processes. Processes that can be ordered in chains of events, such as the chain of dispersion, deposition, chemical buffering in the soil, and biotic response. In addition to following such chains, we can distinguish simultaneous processes with similar effects, such as the buffering by the solution of $CaCO_3$, the weathering of silicates, the exchange of cations, and the decay of aluminium and iron (hydr-)oxides (after Ulrich, 1980; Verstraten, 1982; De Vries & Breeuwsma, 1986).
Now, if we would like to have structural characteristics for our classification, because these are easier to measure or to recognize, we could relate the processes to so-called controlling factors: structural characteristics that control the processes. However, since we want to cover a large number of environmental hazards, ranging from climatic change to disturbance by noise, we would have to define all the parameters relevant for all the processes related to all environmental hazards. Thus, we might be drowned in complexity.

Still another complicating factor must be mentioned. It concerns the fact that some important environmental problems have global dimensions, such as climatic change, whereas others are confined to certain regions, such as overgrazing. It is obvious that it is

[1]This chapter is based on revised texts from Klijn, F. & H.A. Udo de Haes, 1994. A hierarchical approach to ecosystems and its implications for ecological land classification. *Landscape Ecology 9: 89-104*, and F. Klijn, 1994. Spatially nested ecosystems: guidelines for classification from a hierarchical perspective. Pages 85-116 in Klijn, F. (ed.), 1994. Ecosystem classification for environmental management. Kluwer Academic Publishers, Dordrecht.

extremely difficult, if not impossible, to develop a classification which is equally appropriate for problems at the global scale, at the local scale, and at all intermediate scales at the same time. And even if such a classification were possible, we would still be confronted with the practical problem of mapping the relevant characteristics at such divergent scale levels. We should therefore perhaps first decide on the spatial scale level, which would then guide the classification.

We have thus illustrated that classifying ecosystems is complicated owing to the fact that ecosystems are intricate systems, which results in a large number of possible approaches. In order to know where to begin, a number of decisions must be taken. We might summarize these in terms of questions and answers. Firstly, is the restriction to structural characteristics the answer? Only partly, it seems. Secondly, is deciding on the purpose of the classification the answer? Again, apparently only partly. And thirdly, is the spatial scale level the decisive factor? Again, we think not entirely.
In fact, we are convinced that we can only find our way out of this maze by an approach which is both holistic and deductive. In other words, we argue for an approach from a theoretical basis.

4.1.1 Inductive and deductive approaches to classification

A principal point of difference in approaches to classification is the difference between induction from data and deduction from theory.
In an *inductive* approach priority is given to the sampling of data followed by applying relatively standard quantitative methods to arrive at a classification (Bunce & Heal, 1984; Bunce, 1994), *e.g.* TWINSPAN (Hill, 1979) or suchlike. It starts without any postulate about relations, and assumes that similarities and dissimilarities will emerge from the data. This approach is the common approach in plant sociology (e.g. Schaminée et al., 1995), but is also used in integrated ecological land classification, either more implicitly by starting with aerial photo-interpretation (Zonneveld, 1972; 1995; Van Gils, 1989) or more obviously by submitting geographical data of various kinds to TWINSPAN-clustering (*cf.* Bunce, 1994; Kuivenhoven & Salden, 1995).
A *deductive* approach starts with a set of hypotheses of how nature is built up and functions. For ecosystem classification such hypotheses could imply a notion of what causes the differences between ecosystems. This allows focusing on these supposedly differentiating factors from the outset onwards, and results in a relatively strict guideline for classifying and mapping (*cf.* Bailey, 1987; 1996). Of course, the hypotheses are based on previous empirical research and experience. The main scientific task of this approach is testing the hypotheses. Starting from a general conception, the theories and concepts are recurrently adapted after trials.

It is often supposed that an inductive approach is objective in contrast to a deductive approach, which obviously relies on subjective choices. However, clustering according to an inductive approach also relies on many subjective choices, because the outcome of any inductive classification is determined primarily by the sample (Bunce *et al.*, 1981; Bunce, 1994; Runhaar *et al.*, 1994): its size, the way it is stratified and the characteristics taken into account all depend on subjective choices. The effect of undersampling or oversampling some ecosystems on the resulting classification must not be underestimated (see, for

example, Runhaar *et al.*, 1994). In addition, the choice of ecosystem characteristics surveyed affects the outcome (*cf.* Kuivenhoven & Salden, 1995).
Thus, no classification, whether deductive or inductive, can be obtained without making a number of subjective choices. This is a principal dilemma of classification *per se*. Principly, for the classification and mapping of complex objects such as ecosystems or ecological land units, a deductive approach with subsequent testing need be no more arbitrary than an inductive approach from data.

We shall follow a deductive approach in this thesis, because of its obvious relation to theories about ecological relationships, because of its expected general validity, and because of the resultingly large applicability for extrapolation. We shall argue this choice.
With an inductive approach one necessarily remains uncertain about the outcome, which shows up only in the end. This outcome may or may not be consistent with earlier findings, it may be trivial, or it may be un-interpretable. In contrast, a deductive approach is closely related to (most or preferably all) earlier findings from the beginning onward, as these earlier findings are the source of the hypotheses about ecological relationships, *i.e.* theory.
Connected with this is the experience that a good theory allows application in a variety of cases, without having to start from scratch each time (*cf.* Lenz, 1994a). It may be expected to have general validity, or at least more general validity than merely for the case under consideration. This allows attention to be focused on deviations from the expected.
Because a deductive approach is based on theories about ecological relationships and has general validity, the results of such an approach may be relatively safely extrapolated to other, adjacent areas and other space and time frames. In contrast, most inductive approaches yield no explanatory theories, limiting their applicability to the case in question (*i.e.* sample area and sample variables).

4.1.2 Procedure

Our deductive approach will be based primarily on hierarchies and includes the following steps:

- postulating a simplified hierarchical model of an ecosystem comprising all relevant components as functionally related subsystems, in order to structure the complexity;
- defining a guideline for the classification of hierarchically nested ecosystems based on this hierarchical model;
- classifying and mapping ecosystems at different spatial scale levels in order to obtain ecosystem maps, which are related to one another both structurally and functionally;
- zooming in on the most relevant processes and parameters for specific applications.

In the next section we shall go into the ecosystem model. Subsequently, we shall explain the guideline for classification based on this as the 'pièce de résistance'.

4.2 A comprehensive hierarchical ecosystem model

A simplified ecosystem model can be based on ideas concerning interactions between the main spheres of our planet (Van der Maarel, 1976; Van der Maarel & Dauvellier, 1978). This spheric model can be transformed into a hierarchical model of an ecosystem which encompasses all the abiotic and biotic components that are ecologically relevant, irrespective of temporal or spatial scale (Bakker *et al.*, 1981; Piket *et al.*, 1987). An adapted version (Klijn, 1988) is presented as Figure 4.1. It can be regarded as a general, hierarchically structured model, which is principally conceptual. It is based on an aggregation of the various abiotic and biotic constituents of ecosystems into a few *major components* or '*subsystems*'. These components relate to one another in a manner representing a rule rather than a specific case.

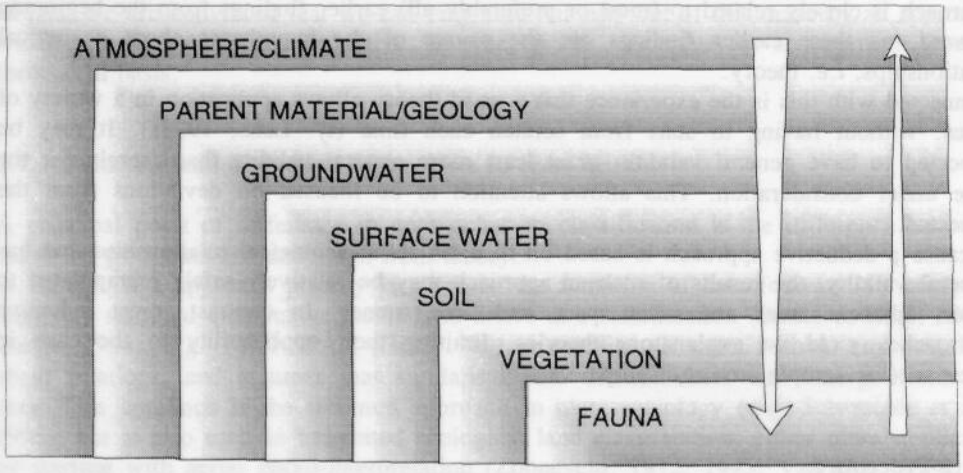

Figure 4.1 Hierarchical model of an ecosystem, showing a hierarchy of relative dependence between the major components (after Van der Maarel & Dauvellier, 1978; Bakker *et al.*, 1981; Piket *et al.*, 1987).

At first glance, the model may appear to be a somewhat deviating conception of an ecosystem, because it contains so many abiotic components. Ecosystems are often considered to be confined to the network of relationships among biota and the directly ecologically relevant variables (*cf.* Leser, 1991, page 39-40). In the case of a plant community such variables would be moisture availability, nutrient availability, acidity and others. However, such an approach is too narrow for environmental management or integrated environmental sciences. Since, for example, susceptibility assessment requires data on abiotic characteristics, the abiotic environment should be incorporated fully into the model. Another reason to take the abiotic components fully into account in a general purpose model is that at global scales the distribution of ecosystems is controlled almost entirely by abiotic factors.

This said, some components included in the model may require some further explanation, especially surface water, which is distinguished separately from groundwater, and soil, which is included as separate component as well. In general, of course, the number and kind of components distinguished is a matter of definition and choice.
For a geologist, surface water may well be regarded as a type of bedrock, like loose sediments, *i.e.* as parent material. Outside this discipline, however, surface water is recognized as a concrete volume with both depth and spatial extension. In this sense it also differs from groundwater, which is intricately connected with a solid parent material. The immediate recognizability of these components led us to distinguish them.
The recognition of soil as a separate component as well as its hierarchical position above vegetation and fauna is also related to a matter of definition. It is often maintaned that, in a strict sense, one may only speak of soil when the parent material is somehow biologically altered, or at least rooted or otherwise invaded by biota. In contrast, many purely physico-chemical processes are regarded as true pedogenetic processes, such as the formation of new clay minerals from the soil solution, clay-illuviation, the leaching of cations, a reallocation of iron, etc. In addition, for soil classification, the parent material (C-horizon and/or R-horizon) is generally fully taken into account in order to permit classification of soils without any pedogenetic alteration, such as Fluvisols (unaltered fluvial sediments), Lithosols (unaltered hard rock) or Arenosols (often unaltered eolian sediments of dune areas). In practice, soil *sensu lato* integrates the influences of at least the above components, *viz.* climate, parent material and hydrological regime, whereas it may be altered by vegetation and fauna to varying degrees.

4.2.1 Hierarchical relations

The conceptual model presented above is hierarchical and consequently related to '*hierarchy theory*'[2] (O'Neill, 1988; Jørgensen, 1992; Müller, 1992), nowadays popular in ecology (*cf.* also Urban *et al.*, 1987). Its hierarchical character is manifold. First and foremost, it expresses that the lower components are relatively dependent on those above, as indicated by the downwardly directed arrow in Figure 4.1. However, this does not mean that one can neglect any reverse influence of the lower components on the upper ones, as is indicated by the upwardly directed arrow. But it also reflects hierarchies of volume, time of evolution and change, direction of fluxes of energy and matter, and patterns of nested systems as well.
All these hierarchies can be ordered in two categories, namely hierarchies of structure and hierarchies of processes. To exemplify this, we summarize a number of observations on hierarchies by Bakker *et al.* (1981) and others, arranging them according to the two categories.

Hierarchies of structure:

- reservoirs diminish in size (parent material > water > soil > vegetation > fauna);
- patterns of the upper components are reflected in the lower ones (climate > soil;

[2]In fact, hierarchy theory is not a true theory, as it appears to have no predictive value. The recognition of hierarchies serves mainly as an ordering tool (cf. also J.A. Klijn (1995).

climate > vegetation; parent material > vegetation).

Hierarchies of processes:

- energy transport is commonly directed downwards;
- matter transport is generally directed downwards;
- the genesis of the lower components is determined by the upper ones (e.g. wind > dunes > sandy soils);
- the existence of lower components depends on the upper components (e.g. parent material > porosity > groundwater);
- changes in the relatively independent components have unavoidable effects on dependent components (e.g. climatic change > surface water discharge > soil erosion > vegetation).

An often expressed critique to the recognition of such a component hierarchy is the notion that in reality all components are always ecologically relevant to some extent, and that all can be studied at whatever spatial and time scale. This may be illustrated by climate, for example, which is an important ecological factor causing zonation with latitude, but is also important in mountainous areas in relation to altitude and exposure, and a significant factor for the herb layer in vegetations with different vertical structure (height) as well. However, from an integrated ecosystem point of view we argue that only in the first case should climate be regarded as the cause of the differences we observe, whereas, in contrast, in the second example geomorphology (*i.e.* altitude and exposure) and, in the third, vegetation cover should be regarded as such. Therefore, we should be continually aware of the causal dependence between the components and arrange the components accordingly.

Even then, many examples may be thought of which contradict the 'general' hierarchical order. For example, for large aquatic systems, such as marine systems, the atmosphere may depend on the water to a greater degree than *vice versa*, whereas the significance of the submerse parent material may be very limited to non-existent in comparison to the water body itself. Dispute may also arise about the position of soil in the hierarchy. It is placed above vegetation, because, in the short term, soil characteristics largely control the possibilities for species, whereas, in contrast, a continuous coverage with vegetation may profoundly alter the soil in the long term only. Consequently, the hierarchical model remains conceptual by definition and, as the above examples show, it remains also sensitive to notions and definitions.

4.2.2 Hierarchies of scale

The aforementioned two categories of hierarchies also relate to spatial and temporal scales. We shall briefly elaborate on these scale aspects, because they form the foundation for the guideline for classification and mapping we shall derive from it.

As to *spatial scales*, we may observe that there is a difference in the spatial scale on which ecosystem components cause patterns on the earth's surface, predominantly reflected by the distribution of biota (*e.g.* Leser, 1976; Walter, 1979; Bailey, 1987; Godron, 1994). We may recall that climate zones, for example, are a global phenomenon determined mainly by latitude. Geological processes and lithology determine the pattern

of mountain ranges and valleys at large scales, thus influencing relief, hydrology and soil formation. Soils show more fine-grained patterns, whereas vegetation superimposes an even finer pattern of various succession stages. In fact, the patterns on the earth's surface we observe belong to the hierarchies of structure mentioned above[3].

As to *temporal scales*, we may consider the natural rate of change of the various ecosystem components. Natural climatic change generally takes tens of thousands of years. Geological and geomorphological processes such as mountain-building, weathering or meandering of rivers need as much time, but some processes may occur in decades, and disasters even within a day. Soil characteristics may change over thousands of years or centuries, although erosion may cause soil degradation even within a few hours. Vegetation may react within a year, although natural succession generally takes some decades or even centuries. Fauna, of course, is the most rapidly responding component in ecosystems: it may simply fly away. These differences in the temporal scale of natural processes are an expression of the hierarchy of processes. This also accounts for the time lag which is so often encountered in the ecological effects of human-induced processes, implying abiotic changes first, such as acidification or climatic change.

These temporal and spatial aspects of the components of the hierarchical ecosystem model are visualized in Figure 4.2 in a simplified way.

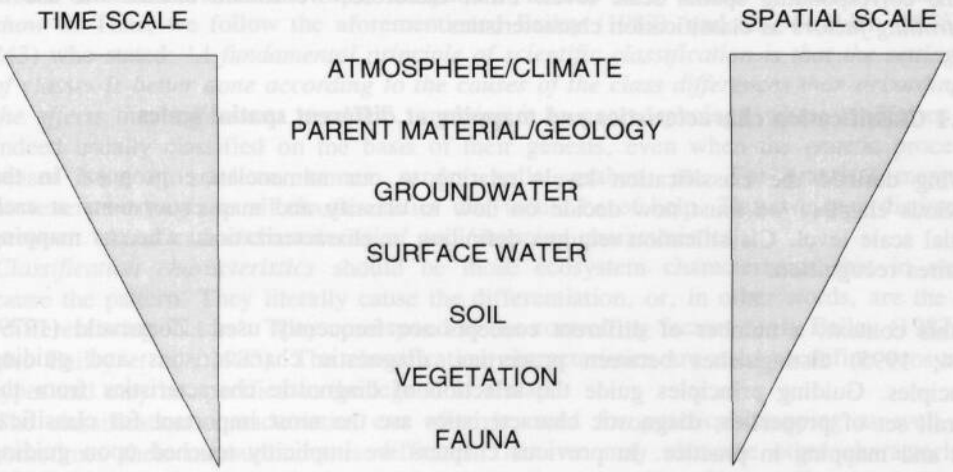

Figure 4.2 The relationship between ecosystem components and spatial and temporal scales.

[3]Of course, patterns can be regarded as being the result of processes. On the other hand, patterns and other structural characteristics influence the various processes. In ecological terms, one might say that there is such an intricate relation between structural and functional characteristics that the distinction becomes somewhat artificial.

4.3 From the hierarchical model towards a guideline for classification

Having postulated the hierarchical ecosystem model, we are confronted with the problem of classifying ecosystems with regard to all their relevant components, as well as to various spatial scale levels. This is practically impossible of course, but we may find a way out based primarily on the recognition of '*correlative complexes*' (Kwakernaak, 1982).
The term 'correlative complex' implies that the whole can be sufficiently known through only certain of its characteristics, owing to the fact that many characteristics are highly correlated. In other words, some ecosystem characteristics are diagnostic for the whole. If we select the most relevant characteristics at a certain spatial scale level as classification characteristics, we can estimate quite a number of other ecosystem characteristics with sufficient accuracy, especially the dependent ones. Consequently, for a practical classification we need not take into account all ecosystem characteristics, nor all components. Of course, these correlative complexes exist only by the grace of the functional relationships between the hierarchically arranged ecosystem components, as depicted in Figure 4.1.

The question thus arises how to define what are the most appropriate classification characteristics at a specific spatial scale level and how to ensure a proper 'functional' connection between the various classification levels. We presume that the relationship between the hierarchy of ecosystem components and spatial scale levels can be used as a *guideline*, provided we base the various classifications on factors that control the pattern at the corresponding spatial scale level. First, therefore, we should discuss the use of *controlling factors* as classification characteristics.

4.3.1 Classification characteristics and mapping at different spatial scales

Having defined the classification levels relating to our nomenclature proposal in the previous chapter, we must now decide on how to classify and map ecosystems at each spatial scale level. Classification requires definition or characterization, whereas mapping requires recognition.

In this context, a number of different concepts are frequently used. Zonneveld (1979; 1994; 1995) distinguishes between properties, diagnostic characteristics and guiding principles. Guiding principles guide the selection of diagnostic characteristics from the overall set of properties; diagnostic characteristics are the most important for classification and mapping in practice. In previous chapters we implicitly touched upon guiding principles *sensu* Zonneveld, *viz.* an 'ecological guiding principle' (*cf.* Zonneveld, 1994, page 27) in Chapter 1 when requiring the classifications to be ecological, a 'functional guiding principle' (*cf.* Zonneveld, 1994, pages 28-29) in Chapter 2 when requiring the classifications to be practical tools for policy analyses, and a 'genetic guiding principle' (*cf.* Zonneveld, 1994, pages 27-28) in Chapter 3 when stressing the topological relationship between ecosystem components at different spatial scale levels. In the current context, the concept of *diagnostic characteristics* is the most important, as these are used primarily for the practical mapping and subsequently also for defining the legend units. As the diagnostic characteristics are used mainly for mapping, they must be macroscopically

recognizable characteristics.
Vink (1980, page 52; also 1983, pages 86-87), in addition to Zonneveld, distinguishes between general landscape characteristics, differentiating characteristics and other diagnostic characteristics. The *differentiating characterictics* are a specific category of diagnostic characteristics used for defining the mapping units, whereas the other diagnostic characteristics provide for a more elaborate characterization of the legend units without being sufficiently homogeneous throughout the mapping units to apply as differentiating characteristics.
The term 'differentiating' *sensu* Vink obviously relates to the process of making distinctions in a classification. This does not necessarily relate to factors causing differences in reality. For example, the species composition of various natural forest stands may well be used for classification purposes as it is easy to establish, but the differences are most likely determined by the combined influence of site factors, such as topoclimate, recurrent fire, soil or groundwater. In our opinion, these controlling factors, as Bailey (1987) calls them, may be regarded as ecologically the most important characteristics. However, they cannot all be recognized equally directly, as is obvious for topoclimate. It is much easier to establish the species composition as an indicator for climatic differences than to separately measure all the relevant climatic variables over many years.

Now, by more strictly distinguishing between classification and mapping, or, in other words, between definition and recognition, we may formulate an alternative approach to both Zonneveld's and Vink's. In our opinion, the most logical is to define classes by means of characteristics that *cause* the pattern, instead of by means of characteristics that *show* it. Thus, we follow the aforementioned Bailey (1987), and also Strahler (1975, p. 243) who stated: '*A fundamental principle of scientific classification is that the setting up of classes is better done according to the causes of the class differences than according to the effects that differences produce*', applying it to geomorphology where landforms are indeed usually classified on the basis of their genesis, even when the genetic processes ceased long ago: eskers, kames, sandrs, sølle, graben, etc. For practical mapping, however, we may use all characteristics that would be of help. Thus we may distinguish between 'classification characteristics' and 'mapping characteristics'.
Classification characteristics should be those ecosystem characteristics that in reality cause the pattern. They literally cause the differentiation, or, in other words, are the real 'differentiating factors'. They correspond to the controlling factors *sensu* Bailey (1987; *cf.* also Bailey *et al.*, 1985). The classification characteristics are used to define ecosystem types and may be specified in the legend of a map.
The classification characteristics can often be used for practical mapping as well, but sometimes their measurement is difficult. Then we may also use other characteristics which adequately reflect the pattern caused by the controlling factors. For this larger set of characteristics which may all — in combination — apply for practical mapping, we use the term *mapping characteristics*.

As we have taken ecosystems as the object of classification and interpreted these rather broadly (see Section 4.1), the *controlling factors* mentioned above are not external factors, but internal to the system. Also the use of the term *factors* is not inhibiting their use, as this is merely a 'functional' equivalent of ecosystem *characteristics*. Soil texture is a characteristic, but at the same time a factor for soil moisture; soil moisture is a

characteristic, but at the same time a factor for plant growth; etc. Only characteristics without ecological relevance are no factors, but then these are also irrelevant for ecological or environmental research. This implies that there are no conceptual obstacles to using controlling factors as classification characteristics.

In fact, this 'genetic' approach is followed not only in geomorphology (Strahler, 1975), as referred to above, but is widely applied. The best known example is probably the scientific taxonomical classification of species, in which taxa at different levels are classified according to genetic relations, whereas determination, obviously, is still based largely on morphology. Here we see the same distinction between classification in the strict sense and determination.

In soil classification too, a genetic approach is often followed, distinguishing on the basis of, for example, podzolisation (Podzols) or eluviation and illuviation of clay (Luvisols), although soil classification sometimes combines a genetic approach with a more morphometric approach because of foreseen applications of the soil classification and mapping.

4.3.2 The component hierarchy as a guideline

The distinction between classification and mapping characteristics enables us to use the relationship between the hierarchy of ecosystem components on the one hand and the classification levels related to spatial scales on the other as a guideline for selecting classification characteristics. This is possible, because the component ranking reflects a hierarchy of dependence, *i.e.* a hierarchy of controlling factors. Reduced to its essentials, the guideline implies that we should use classification characteristics that determine the pattern at the scale level concerned.

This classification guideline is given in Figure 4.3, which visualizes the relationship between determining ecosystem components and scale levels. The ranking of ecosystem components in this figure is the same as in the hierarchical model, whereas the spatial scale levels have been indicated with the corresponding terminology. 'Parent material/geology' has been divided into 'parent material' and 'geomorphology', the combination of which is sometimes addressed as physiography. Geomorphology (relief) cannot be regarded as a component because it has no volume, but is such an essential controlling factor that it deserves a separate place.

We may now query how this guideline stands in a practical context. To this end, we shall look more closely at the characteristics used for classifying and mapping ecosystems or individual ecosystem components at different spatial scale levels in practice.

We then encounter striking resemblances in pattern. The zonal differentiation of soil groups (FAO, 1988) correlates perfectly with vegetation zones (Walter, 1979). At the spatial scale of continents, we can observe a reflection of physiography in both soil pattern and vegetation pattern, and so on. These observations may be explained as follows. Although it may have been the intention to map soils or vegetation at global scales, the pattern which results on the map reflects the controlling factors, because vegetation and soil are only indicative of climatic differences at this spatial scale. In other words, the classification characteristics that have been defined in the legend are not the same as the characteristics which apparently cause the pattern. It would have been better to call the maps, respectively, an ecoclimatic or bioclimatic map, and a pedoclimatic

map. The legends, *i.e.* the classifications, should be adapted correspondingly. In our opinion, this would be more straightforward.

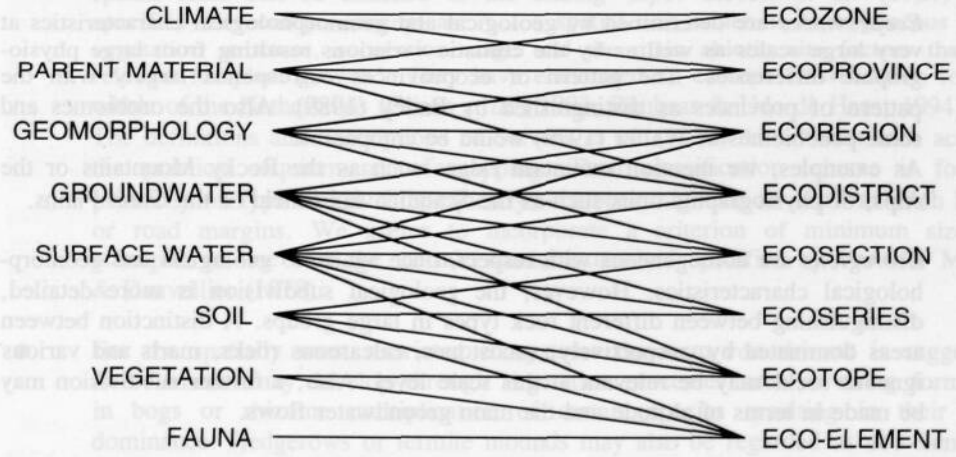

Figure 4.3 The relationship between spatial scale levels (classification levels) and ecosystem components proposed as a guideline for the selection of classification characteristics.

Summarizing the above, we argue that a classification of ecosystems should be performed at different spatial scales by focusing on those components that determine the pattern at a certain scale level. Thus we can use the generalized hierarchical ecosystem model as a guideline for selecting classification characteristics. On the other hand, the practical mapping may be based on all characteristics that might be of help for recognition, either on remotely sensed images or in the field.

4.3.3 Brief characterization of the various classification levels

Above, we have formulated a guideline for selecting the most appropriate classification characteristics for different spatial scale levels. Below, we shall specify the various classification levels in general terms, following the nomenclature proposed in the previous chapter.

- *Ecozones* are related to the world-embracing climate zones. They can be distinguished on the basis of, for example, the Köppen system for regional climates (Trewartha, 1968). The pattern is reflected in the zonal pattern of soils (Dokuchaev Soil Institute, 1963; FAO, 1988) and vegetation (Walter, 1979). Hence, ecozones correspond largely with the zonobiomes of Walter (1979) or the domains and some divisions of Bailey (1989). The main ecozone types may be indicated as

the arctic, the subarctic with tundra vegetation as main diagnostic characteristic, the boreal zone with taiga coniferous forests, the temperate zone, the mediterranean, the (semi-)deserts, the dry tropical zone with savannas and the wet tropical zone with rain forests.

- *Ecoprovinces* are determined by geological and geomorphological characteristics at very large scales as well as by the climatic variations resulting from large physiographic differences. The pattern of ecoprovinces corresponds largely with the pattern of provinces as distinguished by Bailey (1989). Also the orobiomes and some pedobiomes of Walter (1979) would be comparable.
  As examples, we mention mountain ridges such as the Rocky Mountains or the Alps, or physiographic units such as the Scandinavian Shield or the Great Plains.

- *Ecoregions* are homogeneous with respect, once again, to geological and geomorphological characteristics. However, the geological subdivision is more detailed, distinguishing between different rock types in large groups. A distinction between areas dominated by, respectively, sandstones, calcareous rocks, marls and various igneous rocks may be relevant at this scale level. Also, a further subdivision may be made in terms of altitude and the main groundwater flows.

- *Ecodistricts* are spatial units that are homogeneous as to slowly changing geological, geomorphological, groundwater and surface water characteristics. These correspond largely with soil groups as determined by the parent material. At this scale, we can distinguish physiographic units such as large valley systems with Fluvisols, individual volcanoes with Andosols, deltas with Inceptisols or salt plains with Solodic Planosols.

- *Ecosections* are spatial units which are homogeneous with respect to individual but large geomorphological features, such as lowland brook valleys, divides, individual slopes, slumps, mud-flows, debris cones, etc. They are distinguished on the basis of geomorphology, soil groups and groundwater regime as controlled by geomorphology: leaching or upward seepage.

- *Ecoseries* are homogeneous with respect to characteristics of soil, groundwater and surface water, which can be established in the field with simple instruments and devices such as a hand auger. The site characteristics used for classifying ecoseries should preferably be stable for periods of tens of years. This almost automatically implies a kind of ecological soil classification, because the soil can be considered the most important site factor for plant growth in most parts of the world (see Chapter 8).
  Within ecoseries, vegetation types can be encountered which differ in vegetation structure. These can be understood as succession stages, or semi-permanent stages due to land use practices, thus forming series of vegetation types related in time. Hence, an ecoseries classification and mapping is especially valuable for estimating potential or climax vegetations or for forecasting vegetation developments (see Chapter 9).

- The term *ecotope* is one of the oldest in landscape ecology (Tansley, 1939; Troll, 1968; see also Neef, 1967; Haase, 1973; 1989; Leser, 1991). Use of this concept signifies reference to ecosystems of a certain dimension, such as a raised bog, an agricultural field, a small forest patch or a dune slack grassland. Hence, a certain spatial scale can be attached to the ending *-tope*. Stevers *et al.* (1987) have specified the ecotope concept as: "... a spatial unit which is homogeneous as to vegetation structure, succession stage and the main abiotic site factors that are relevant for plant growth". They have elaborated an ecotope classification for the whole of the Netherlands (Groen *et al.*, 1993; Runhaar & Udo de Haes, 1994).
  The definitions and descriptions of Stevers *et al.* (1987) do not take into account the practical requirements of mapping. Their classification scheme is focused primarily on typification. Consequently, ecotopes might be as small as ditch banks or road margins. We prefer to incorporate a criterion of minimum size for ecotopes, in line with the 'certain spatial extension' specified by Van der Maarel & Dauvellier (1978).

- For the spatially most restricted ecosystems the term *eco-element* is suggested. Eco-elements may develop through vegetation processes such as string formation in bogs or rhizome multiplication of certain species resulting in their local dominance. Hedgerows or termite mounds may also be regarded as eco-elements. These small-scale ecosystems can be mapped as patches, but only at very detailed mapping scales. In most instances their location will be indicated on maps by means of symbols.

4.4 Summary

Since ecosystems are very intricate systems that can be classified by means of many different characteristics, it is difficult to decide on where to start. We have argued that only a holistic and deductive approach, *i.e.* top-down from a theoretical basis, can lead us out of the maze.

This approach is founded on a simple and general conception of an ecosystem. To this end, the ecosystem is depicted by means of a generalized hierarchical model with the main ecosystem components arranged according to relative dependence and a number of other hierarchies.

Instead of attempting to develop one systematic hierarchical classification scheme for all scale levels, we hold the opinion that a spatial hierarchy requires a different approach: the classification of spatially nested ecosystems can be satisfactorily performed only by developing a series of related ecosystem classifications.
Each classification should be specific for a pre-determined spatial scale level, and should preferably be functionally related to the next scale levels above and below. The relationship with the classifications at adjacent scale levels should be based on dependence relations between the ecosystem components.

We have argued that the classifications should be based on controlling factors, *e.g.* climatic and geological characteristics for large spatial units, and characteristics of soil

and vegetation for the smallest spatial units. Thus, we can use the hierarchy of ecosystem components as a guideline for selecting the most appropriate classification characteristics, because each component influences the pattern on the earth's surface at a different scale level.

In Figure 4.3 this guideline was depicted by relating the hierarchy of ecosystem components to ecosystems at various spatial scale levels indicated with the nomenclature proposed in the previous chapter.

Since the so-called controlling factors determine the pattern on the earth's surface, they should be used as *classification characteristics* to define the classes or legend units. In contrast, for practical mapping purposes all characteristics apply that are useful for recognition; these we call *mapping characteristics*.

5 Ecoregions and ecodistricts of the Netherlands[1]

5.1 Introduction

In the previous chapter we gave only a brief description of the various classification levels. In this chapter we apply the guideline for classification and mapping to the classification levels of ecoregions and ecodistricts, with special emphasis on their practical mapping for the Netherlands. This chapter illustrates the practical obstacles encountered in such an enterprise and the deviations from theory that are sometimes required for practical reasons.
Before moving on to the core of this chapter, the practical purpose of and consequent requirements for the ecoregion and ecodistrict classifications and maps will be introduced.

5.1.1 On the purpose of ecological regionalization

'*State-of-the-environment reports*' provide data on environmental problems, their causes, their effects and present environmental quality. In these background documents for environmental management, which for the Netherlands are regularly compiled by the National Institute of Public Health and Environmental Protection (RIVM, 1989; 1991; 1994), various integrative frameworks are used to present information in an accessible and comprehensive way. Since 1988, regional integration has been an important integrative framework (RIVM, 1989). Within this framework, actual and foreseen environmental problems are investigated for individual areas, against a setting of the natural characteristics of these areas.

For the purpose of this regional integration we mapped the Netherlands' ecoregions and ecodistricts as a first practical application of the classification framework presented in the previous chapter (Klijn, 1988; Klijn & Udo de Haes, 1994). These two spatial scale levels best meet the practical requirements for use at the national level.
The main motive for the mapping was the notion that many different geographical regionalizations were used in environmental research and management, which was especially confusing for policy-makers. Therefore, the Ministry of Housing, Physical Planning and Environmental Management (VROM) commissioned us to either select or establish a standardized regionalization for the Netherlands' environmental management (Klijn & Koster, 1988).

The main requirements for this regionalization were:

- that it should be sufficiently general to be relevant for national environmental management;
- that it should have no more nor fewer boundaries than strictly necessary;

[1]This chapter and the next are slightly revised versions of halves of Klijn, F., R.W. de Waal & J.H. Oude Voshaar, 1995. Ecoregions and ecodistricts: ecological regionalizations for the Netherlands' environmental policy. *Environmental Management* 19/6: 797-813

- that it should not change too rapidly with time, remaining valid for several decades;
- that it should be ecological, especially with respect to differences in susceptibility to acidification, eutrophication, pollution with toxicants, and drought damage or 'desiccation'[2];
- that it should be relevant for all land use functions, including nature conservation, to permit integrated environmental quality assessment;
- that it should permit sensible aggregation of the results of sophisticated predictive modelling and scenario studies on different kinds of environmental management, in order to enhance communication with policy-makers by confronting them with maps referring to recognizable and well-known geographical units;
- and that it should provide a sound basis for stratifying sampling for the purpose of monitoring environmental quality.

Comparable ecological regionalizations are known from the United States and Canada (Rowe & Sheard, 1981; Bailey, 1983; Bailey *et al.*, 1985; Omernik, 1987; 1995), in use for similar purposes (Klopatek *et al.*, 1981; Bailey *et al.*, 1985; Hughes & Larsen, 1988; Hughes *et al.* 1990; Hunsaker *et al.*, 1990; Clarke *et al.*, 1991).

5.1.2 Some prior considerations

In developing an ecological land classification, we first have to decide on the mapping scale. This depends primarily on the purpose of the mapping and the policy level for which the map is intended, as we have argued that the subject matter of the mapping, ecosystems, is not restrictive in this respect. Obviously, a different level of detail is required for national environmental management than for the local level. In the present case, we were obliged to compile a map that would fit on an ordinary page in State-of-the-environment reports. This meant a mapping scale of about 1: 1,000,000 to 1: 5,000,000.

In the previous chapter we argued that for ecological land classification we should define the classes by characteristics that *cause* the pattern on the earth's surface, and not on characteristics that *reflect* it. However, for the practical mapping we might use any ecosystem characteristics, whether they cause a pattern or merely reflect it. The main requirement for these mapping characteristics was that their pattern should be easily, *i.e.* macroscopically, recognizable, preferably even by remote sensing.
Consequently, for the *classification* of ecoregions and ecodistricts we are mainly interested in factors that control the pattern at a mapping scale of about 1: 1,000,000 to 1: 5,000,000 (Figure 5.1). Ecoregions should then be classified by geological and geomorphological characteristics, *i.e.* physiography. For larger water bodies characteristics of the surface water may also be used, such as the distinction between salt tidal and fresh inland

[2]Drought damage and 'desiccation' are both inadequate translations of the Dutch '*verdroging*'. '*Verdroging*' is one of the themes of the Netherlands' environmental management, used to indicate a group of related environmental problems resulting from water shortages due to groundwater extraction and enhanced drainage.

waters. Ecodistricts should be classified on the basis of geological and geomorphological characteristics as well, but subdivided into smaller spatial units formed by individual physiographic units such as ice-pushed ridges, large ice-lobe valleys, floodplains, or polders. Geohydrological characteristics may form a suitable additional classification characteristic, allowing the distinctions of main infiltration/recharge and upward seepage/discharge areas.

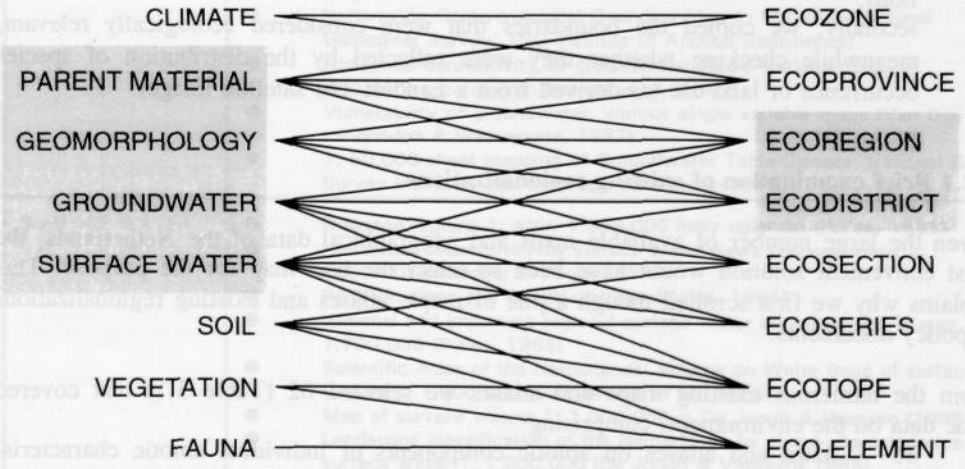

Figure 5.1 According to the guideline the classifications of ecoregions and ecodistricts should be based on geological, geomorphological and hydrological characteristics.

So far for the theoretical guideline. In practice, of course, controlling factors always interact, which implies that a more precise specification of the classification characteristics used can only be given for each legend unit separately. For the ecoregions and ecodistricts of the Netherlands we shall specify this below. Unfortunately, this interaction also complicates the establishment of a really systematic and formal legend. We have tried to evade this problem by using general descriptive terms to indicate the correlative complexes we are talking about: terms of a relatively 'holistic' nature, which contain a wealth of information for those who understand. For example, the term estuary indicates a landform constituted by the interaction of river discharge and tidal influences, in which erosion and sedimentation are currently very active, building sand banks and shoals, with a relatively large intertidal zone where birds may feed and sea mammals may rest, and with only sparse pioneer and closed herbaceous vegetation on salt to brackish sites in contrast to taller vegetations on fresh sites, and with merely initial soil development, *i.e.* ripening, in places that are flooded only during storm surges.

5.2 Method

After these considerations of a theoretical and practical nature, we now return to our primary concern: to either select or develop a comprehensive ecological regionalization for environmental management. We proceeded as follows:

- firstly, we examined existing maps, atlases and existing regionalizations in recent policy documents, in order to select either the optimum existing example (which we did not find), or the most useful ones for compiling our alternative regionalizations;
- secondly, we copied the boundaries that were considered ecologically relevant, meanwhile checking whether they were reflected by the distribution of species occurrence or land use (as derived from a Landsat-TM satellite image).

5.2.1 Brief examination of existing regionalizations

Given the large number of available maps and geographical data of the Netherlands, the most convenient solution would have been to select the best map for the purpose. This explains why we first scrolled though a pile of maps, atlases and existing regionalizations in policy documents.

From the numerous existing maps and atlases we selected 62 (Table 5.1) that covered basic data on the environment, comprising:

- 35 maps and atlases on abiotic components or individual abiotic characteristics, *i.e.* on climate, geology/parent material, relief/geomorphology, groundwater, surface water and soil;
- 15 maps and atlasses on biotic components or individual characteristics, *i.e.* on land use/land cover, vegetation/flora, terrestrial and aquatic fauna;
- 7 integrated maps on the abiotic environment;
- 5 integrated maps on the entire abiotic and biotic environment.

Table 5.1 List of maps and atlases concerning individual ecosystem components as well as partly (abiotic environment) or fully integrated ecological regionalizations that were examined (continued on the next pages).

ABIOTIC COMPONENT MAPS AND ATLASES	
CLIMATE	• Climatic regions 1: appr. 2,000,000 (Van der Maarel & Dauvellier, 1978) • Climate Atlas of the Netherlands (KNMI) • 'Grote Bos' school atlas
GEOLOGY/ PARENT MATERIAL	• Nationwide Geological Overview Map 1: 600,000 (Zagwijn & Van Staalduinen, 1975) • New Geological Map of the Netherlands, subregions 1: 4,000,000 (National Geological Survey) • Simplified Geological Map 1: 2,000,000 after National Geological Survey (Zonneveld, 1985)

GEOMORPHOLOGY	• Nationwide Geomorphological Overview Map 1: 600,000 (National Soil Survey Institute) • 'Lowlands' and 'Uplands' according to Pulles (1985) • Scientific Atlas of the Netherlands, volume on Geology (Zagwijn *et al.*, 1986) with map 1: 1,000,000 • 1: 50,000 sheet mapping (National Soil Survey Institute) (no full coverage)
GROUNDWATER	• Hydrological Regionalization of the Netherlands 1: appr. 2,000,000 (Engelen, 1980) • Nationwide Geohydrological Overview Map 1: 500,000 (National Geological Survey/ TNO Institute of Applied Geoscience) • Horizontal connectivity by groundwater flows (various maps 1: 500,000) according to Roelofs *et al.* (1982) • Vulnerability of groundwater, various single variable maps (Van Duijvenbooden & Breeuwsma, 1987) • 1: 50,000 sheet mapping of Groundwater Table Classes (National Soil Survey Institute) (full coverage achieved in 1995)
SURFACE WATER	• Drainage regions 1: appr. 1,500,000 (only uplands) (Pulles, 1985) • Horizontal connectivity by surface water flows (various maps 1: 500,000) according to Farjon (1982) • PAWN-regions and PAWN-districts (Pulles, 1985) • National and important regional surface water infrastructure 1: appr. 1,700,000 (Pulles, 1985) • Scientific Atlas of the Netherlands: volume on Water (map of surface waters 1: 1,000,000 (Van de Ven *et al.*, 1985) • Map of surface waters 1: 1.000.000 in De Jongh & Vaessen (1985) • Landscape classification of the Netherlands by visual characteristics of surface water 1: 1,000,000 (De Jongh & Vaessen, 1985)
SOIL	• Generalized soil map of the Netherlands 1: 1,000,000 (De Bakker & Schelling, 1966) • Nationwide Soil Overview Map 1: 600,000 (National Soil Survey Institute) • Soil map after Edelman 1: appr. 2,000,000 (Zonneveld, 1985) • Soil map after De Bakker & Schelling 1: about 2,000,000 (Zonneveld, 1985) • Generalized soil map with the most important gradients 1: appr. 2,000,000 (Baaijens, 1985) • Soil map 1: appr. 2,000,000 (Van der Maarel & Dauvellier, 1978) • Natural nutrient status of soils 1: appr. 2,000,000 (Van der Maarel & Dauvellier, 1978) • Soil map 1: appr. 1,500,000 (Pulles, 1985) • Scientific Atlas of the Netherlands: volume on Soil (with map 1: appr. 1,000,000) (Steur et al, 1985) • Nationwide Soil Overview Map ('Nebo') 1: 200,000 (National Soil Survey Institute) • New Nationwide Soil Overview Map ('New Nebo') 1: 250,000 (National Soil Survey Institute) • Generalization of the former for the Nationwide Environmental Mapping (LMK) 1: 200,000 (Kloosterhuis & Pape, 1976; Kalkhoven *et al.*, 1976) • 1: 50,000 sheet mapping (National Soil Survey Institute) (full coverage achieved in 1995)

INTEGRATED ABIOTIC ENVIRONMENT MAPS	
• Physical geographical landscapes of the Netherlands 1: 600,000 (Hof, 1983) • The large natural landscape units 1: 2,200,000 (Keuning, 1965 after Keuning, 1934; *cf.* Hof, 1983) • Physical geographical landscape units 1: 1,300,000 (De Gans & Schoute, 1973; *cf.* Hof, 1983) • Physical geographical provinces of the Netherlands 1: appr. 2,000,000 (Zonneveld, 1985) • Physical geographical landscape types 1: appr. 2,000,000 (Zonneveld, 1985) • Physical geographical landscapes 1: appr. 2,500,000 (Visscher, 1975) • Landscapes with environmental gradients 1: appr. 1,500,.000 (Van der Maarel & Dauvellier, 1978)	
BIOTIC COMPONENT MAPS AND ATLASES	
LAND USE	• Land use statistics (Central Statistics Agency (CBS)) (no overview map) • 4th Forestry Statistics (Central Statistics Agency (CBS)/ National Forestry Service (SBB)) (no overview map) • Mapping of forests, nature reserves and WACs (National Forestry Service (SBB)/ NMF, 1984) • Agricultural regions 1: appr. 1,500,000 (Central Statistics Agency (CBS), 1986)
FLORA/ VEGETATION	• Atlas of the Netherlands' Flora: atlas in 3 volumes concerning species occurrence • Plant Geographical Districts 1: appr. 2,500,000 (in Heukels & Van Oostroom, 1975; Heukels & Van der Meijden, 1983) • Distribution of epiphytes (Barkman, 1958) • Lichen archive of the Netherlands (De Wit, 1976) • National Environmental Mapping, map of nature values (Kalkhoven *et al.*, 1977)
FAUNA	• Breeding bird districts of the Netherlands (Kwak *et al.*, 1987) • Atlas of the Netherlands' breeding birds (Teixeira, 1979) • Atlas of the Netherlands' winter guests and migratory birds (Bekhuis *et al.*, 1987) • Atlas of the Netherlands' butterflies (Geraedts, 1986) • Source and sink areas of aquatic fauna (Vos *et al.*, 1982) • Hydrobiological districts of the Netherlands 1: appr. 3,000,000 (De Lange & Ruiter, 1977; cf Vos *et al.*, 1982)
INTEGRATED ECOLOGICAL REGIONALIZATIONS (ABIOTIC-BIOTIC)	
• National Environmental Mapping, potential natural vegetation (in fact an interpretation of the soil map) 1: 250,000 (Kalkhoven *et al.*, 1977) • Geotopes of the Netherlands 1: 2,500,000 (Visscher, 1972) • Geotope types 1: appr. 1,500,000 after Visscher (1972) (Van der Maarel & Dauvellier, 1978) • Landscape typification and mapping by cultural-historical characteristics according to De Boer, Entrop & Tack (Renes, 1982) • Scientific Atlas of the Netherlands: volume on Landscape with map of Landscapes of the Netherlands 1: 1,000,000 (Piket *et al.*, 1987)	

We assessed these maps and atlases on criteria of applicability, especially in relation to the requirements explained in Section 5.1, namely:

- ease of understanding for professionals and policy-makers
- comprehensiveness of map and legend
- stability over time, required up-dating period

- ecological relevance
- applicability for assessments of susceptibility, significance and vulnerability
- accuracy in relation to policy planning and management measures

The component maps we screened, especially those of abiotic components, were found to be primarily compiled for scientific purposes and not sufficiently applicable for environmental management. They show the distribution of individual components and only seldom take into account the relationships with biota. They generally contain, simultaneously, too much information as a result of their scientific character, and too little because they cover only one component or are derivatives for one well-defined purpose. Moreover, they often have too complex legends, requiring interpretation by specialists. In contrast, component maps have, by definition, the advantage of being relatively unambiguous, because the object of mapping is less complex than 'the total environment'. This makes them very useful sources of information.
For biotic components, we also found that the reliability of species (flora and fauna) and vegetation surveys is insufficient for them to serve as basis for mapping purposes. Moreover, biota are not sufficiently stable over time. Only land use was found to be mapped sufficiently accurately, but the diagnostic value of land use for environmental conditions is considered inadequate.

The relatively large number of integrated abiotic environment maps were found to be largely so-called physical-geographical maps. These are often based strongly on geogenesis and the various examples showed strong similarities.
In contrast, integrated ecological regionalizations, *i.e.* maps that integrate both abiotic and biotic components, were found to be relatively scarce. Moreover, most of these are not so much real integrations of component data, but rather ecological interpretations of integrated abiotic environment maps. When they are indeed based on ecologically relevant abiotic characteristics, they may provide relatively simple maps that are sufficiently applicable to environmental management. One of the most useful examples of such a map was found to be the geotope map of Visscher (1972; see also Van der Maarel & Dauvellier (1978). This map most closely suits our purposes, but it lacks an explanatory legend and does not sufficiently take into account ecologically relevant groundwater characteristics (upward seepage). The map of 'Landscapes of the Netherlands' enclosed in Piket *et al.* (1987) was also considered an inspiring example, though it was found to be biased by an emphasis on ecologically less relevant scenic aspects.

In summary, the screening revealed that not one existing map met all our requirements, although many useful elements could be retrieved from their combination, in terms of both the exact extension and location of relevant spatial units which could be used for the 'mapping', and the wealth of information that could serve as a check on the ecological relevance by checking on correspondences.

5.2.2 Compiling maps from existing geographical data

The results of the screening led us to decide to compile alternative regionalizations, based on a selection of the existing maps. The following sources were regarded as most useful and sufficiently accurate for drawing the maps of ecoregions and ecodistricts:

- the National Geological Survey 1: 600,000 (Van Staalduinen, 1979);
- the Generalized Soil Map 1: 1,000,000 (De Bakker & Schelling, 1966);
- the Soil Map of the Netherlands 1: 250,000 (Steur *et al.*, 1985);
- a generalized soil map with added information on the most significant gradients 1: appr. 2,000,000 (Baaijens, 1985);
- the volume '*Landschap*' of the Scientific Atlas of the Netherlands (Piket *et al.*, 1987); and
- the '*geotopes*' classification by Visscher (1972; included in Van der Maarel & Dauvellier, 1978).

These maps cover features that correspond best with the suggested classification characteristics for ecoregions and ecodistricts. With the exception of the geological map, they do not themselves involve classification characteristics, but rather highly correlated characteristics that may function well as mapping characteristics. The boundaries between individual ecoregions and ecodistricts were copied from the most relevant or accurate of these maps for each specific boundary.

The other maps and atlases, especially those on biotic components, were used to assess the ecological relevance of the boundaries on the first set of maps. For the same purpose we used a composite Landsat-TM satellite image of the entire country. If boundaries are not reflected by land cover or land use at all, doubts may be entertained about their ecological relevance. Of course, decisions on the relevance of boundaries were not taken without ample consideration of (presumed) ecological significance.

5.3 Classification and maps of ecoregions and ecodistricts

The ecoregion classification for the Netherlands has resulted in four terrestrial types and two aquatic types. This makes six types in total (Figure 5.2). These ecoregions are subdivided into ecodistricts, with 26 terrestrial ecodistrict types and 11 aquatic ecodistrict types being distinguished. This makes a total of 37 ecodistrict types (Figure 5.3).

Both the ecoregions and the ecodistricts have been indicated by three means (see the legends to Figures 5.2 and 5.3):

- a code
- a type indication
- one or more topographical names for discrete mapping units

The type indication is concerned with the general character of the area, implying that mainly geological, geomorphological and/or hydrological terms of a holistic nature were chosen. Topographical names are attributed to discrete mapping units. So far, it has not been considered necessary to distinguish between different ecoregions of the same type within the Netherlands. Therefore, the topographical names presently used for ecoregions do not add much to the type indications. In contrast, ecodistrict types often manifest a disjunct distribution, necessitating a distinction in topographical terms. For their names commonly used nomenclature was retrieved from topographical maps.

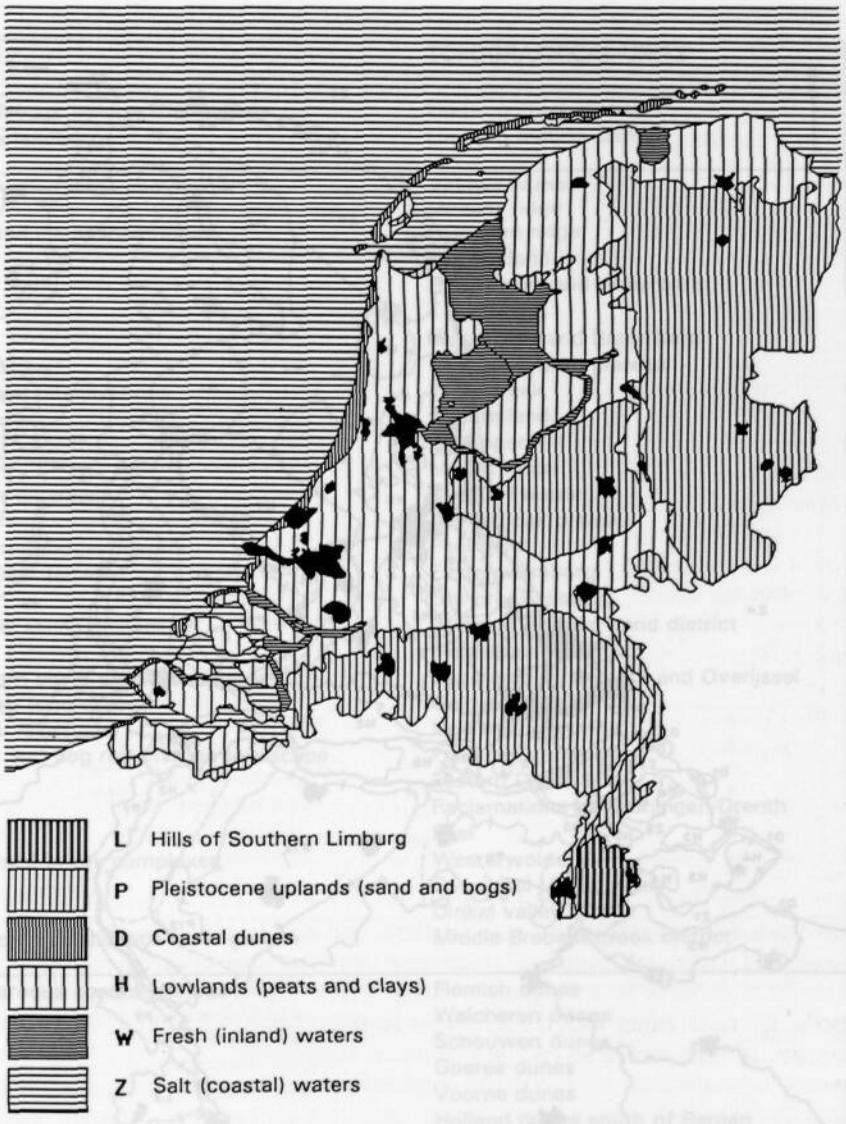

Figure 5.2 Ecoregions of the Netherlands. The order of legend units is such as to take into account horizontal relationships, *i.e.* from high elevation (relatively independent source area) to low elevation (relatively dependent sink area).

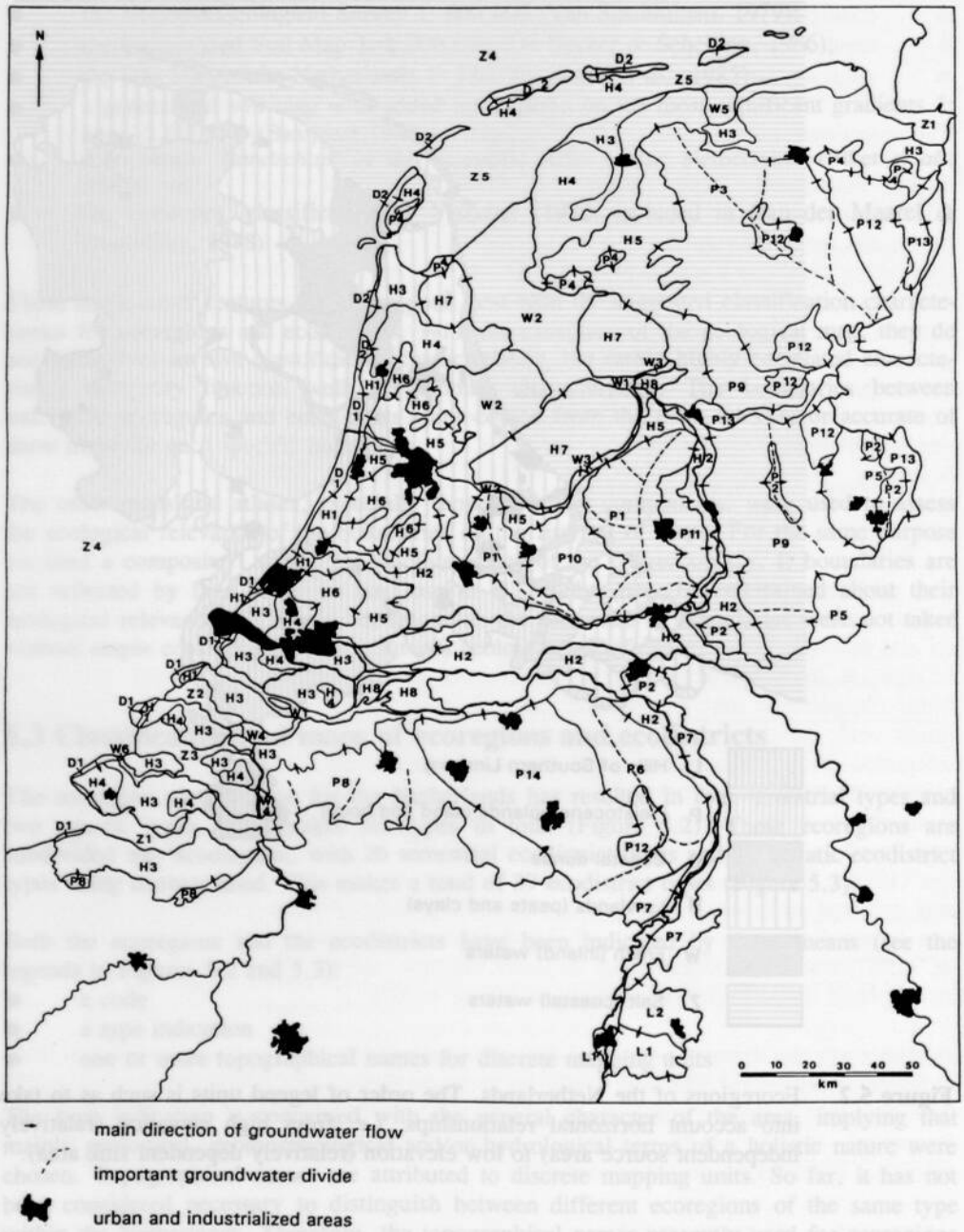

Figure 5.3 Ecodistricts of the Netherlands. The order of legend units is such as to take into account horizontal relationships, *i.e.* from high elevation (relatively independent source area) to low elevation (relatively dependent sink area).

CODE	TYPE	TOPOGRAPHICAL UNITS
L1	Cretaceous hardrock landscape	Limburg chalk district
L2	Loess landscape	Limburg mining district
P1	Ice-pushed ridge complexes	Veluwe Massif
		Utrecht ridge
P2	Isolated ice-pushed ridges	Nijmegen ridge
		Montferland
		Haarlerberg and Holterberg
		Tankenberg
		Kuiperberg and Braamberg
P3	Till plateau	Friesian-Drenth plateau
P4	Isolated till elevations	Woldstreek
		Gaasterland
		Wieringen
		Mount Texel
P5	Other till areas	Middle-Twente
		Achterhoek plateau
P6	Tectonic ridges/ horsts	Peel horst
P7	Old river terraces	Eastern Limburg
		Leu brook district
P8	Fluvial sands landscape	Western Brabant sand district
		Flemish sand district
P9	Eastern cover sands landscape	Sand area of Geldern and Overijssel
P10	Glacier lobe valleys	Geldern valley
P11	Sandr landscape	Veluwezoom
P12	Bogs and bog reclamation landscape	Friesian-Drenth bogs
		Eastern Dutch bogs
		Reclamations of Groningen-Drenth
		Peel
P13	Lowland brook complexes	Westerwolde
		Overijssel Vecht valley
		Dinkel valley
P14	Tectonic subsidence area/ graben	Middle-Brabant brook district
D1	Calcareous coastal dunes	Flemish dunes
		Walcheren dunes
		Schouwen dunes
		Goeree dunes
		Voorne dunes
		Holland dunes south of Bergen
D2	Non-calcareous coastal dunes	Holland dunes north of Bergen
		Texel dunes
		Vlieland
		Terschelling dunes
		Ameland dunes
		Schiermonnikoog dunes
		Rottumeroog
H1	Beach barrier landscape	Dune hinterlands
H2	Floodplain landscape	Meuse valley
		Old IJssel valley
		IJssel valley

		Central Dutch river district
		Old Rhine district
		Utrecht Vecht district
H3	Sub-recent dikings	Dollard polders
		Groningen and Friesian reclamations
		Het Bildt- Middle Sea
		Zijpe
		Westland
		Zeeland and Southern-Holland islands
		Northwestern Brabant
H4	Marine clay inversion landscape	Friesian-Groningen mound district
		Western-Friesia
		South-Holland and Zeeland swamp reclamations
H5	Lowland peats	Friesian-Overijssel lowland peats
		Veluwe north margin
		Eem valley
		Vecht lake district
		North Holland lowland peats
		Heart of Holland
		Krimpenerwaard
		Alblasserwaard and Vijfheerenlanden
H6	Drained lakes	Heerhugowaard
		Beemster
		Purmer
		Wormer
		Haarlemmermeer
		Schieland
H7	Polders	Wieringermeer
		Noordoostpolder
		Eastern Flevoland
		Southern Flevoland
H8	Deltas	Biesbosch
		IJssel delta
W1	Sedimentation basins	Holland Deep- Haringvliet
		Lake Ketel
W2	Fresh inner seas	Lake IJssel
		Lake Marken- Lake IJ
W3	Lakes	Black Lake
		Lake Veluwe-Dronten
		Wolderwijd- Nuldernauw
		Lake Eem- Lake Gooi
W4	Closed estuaries	Lake Volkerak- Zoom
W5	Diked sea intrusions	Lake Lauwers
W6	Brackish lakes	Lake Veere
Z1	Estuaries	Western Scheldt
		Ems- Dollard Estuary
Z2	Saline lakes	Lake Grevelingen
Z3	Sea branches	Eastern Scheldt
Z4	Coastal seas	North Sea
Z5	Wadden seas	Wadden Sea

The classification characteristics considered most determinant for the boundaries of the various ecoregion and ecodistrict types are specified in Tables 5.2 and 5.3, as well as the main and secondary mapping characteristics. These tables reveal the use of different classification characteristics for different ecoregion or ecodistrict types, although they are all from the same range of mainly physiographic characteristics.

Table 5.2 Classification and mapping characteristics as derived from different components for each ecoregion type.

classification characteristics (■■■)
mapping characteristics that are highly correlated with the classification characteristics (■■)
dependent characteristics of additional but minor importance for the mapping (■)

COMPONENT / ECO-REGION TYPE	GEOLOGY/ PARENT MATERIAL	GEOMORPHOLOGY	GROUNDWATER	SURFACE WATER	SOIL	VEGETATION	FAUNA
L	■■■	■■	■■	■	■	■	■
P	■■■	■■	■■	■■	■	■	■
D	■■■	■■	■■	■	■	■	■
H	■■■	■■	■■	■■	■	■	■
W		■■■		■■	■	■	■
Z		■■■		■■	■	■	■

For environmental management not only the *topological* relationships between the components within the recognized ecosystems are important, but also horizontal or so-called *chorological* relationships (Vos *et al.*, 1982; Vos & Opdam, 1993) between ecosystems, because these determine how large an area may be affected by pollution, eutrophication or other environmental hazards. Chorological relationships can be recognized at any scale level. In the present context, we are especially interested in relationships between ecoregions and between ecodistricts. At these spatial scale levels geohydrological and hydrological relationships are the most important, both predominantly driven by gravity. To account for these interactions, we ordered the ecoregion types and ecodistrict types in the legends from high elevation to low elevation. This corresponds with the main direction of groundwater flow and surface water flow from mainly *source areas* (catchment area, infiltration/recharge) to predominantly *sink areas* (sedimentation basins, upward seepage/discharge). In addition, we indicated the main flow directions on the ecodistrict map by means of arrows (after Hof, 1983), as well as a number of important (ground)water divides. The latter give an indication about the size of catchments in the more elevated ecodistricts. Their location was derived from the maps of Roelofs *et al.* (1982) and Baaijens (1985).

Table 5.3 Classification and mapping characteristics for ecodistricts as derived from different ecosystem components for each type.

classification characteristics (■■■)
mapping characteristics that are highly correlated with classification characteristics (■■)
dependent characteristics of additional but minor importance for the mapping (■)

ECO-DISTRICT TYPE / COMPONENT	GEOLOGY/ PARENT MATERIAL	GEOMORPHOLOGY	GROUNDWATER	SURFACE WATER	SOIL	VEGETATION	FAUNA
L1	■■■	■■■	■■	■	■	■	■
L2	■■■	■■■	■■	■	■	■	■
P1	■■■	■■■	■■	■	■	■	■
P2	■■■	■■■	■■	■	■	■	■
P3	■■■	■■■	■■	■	■	■	■
P4	■■■	■■	■	■	■	■	■
P5	■■■	■■	■	■	■	■	■
P6	■■■	■■■	■■	■	■	■	■
P7		■■■	■	■	■	■	■
P8	■■■	■■	■■■	■	■	■	■
P9	■■■	■■	■■	■	■	■	■
P10	■■	■■■	■■■	■	■	■	■
P11	■■	■■■	■■	■	■	■	■
P12	■■■	■	■■■	■	■	■	■
P13		■■■	■■	■■■	■	■	■
P14	■■■	■■■	■	■■	■	■	■
D1	■■■	■	■	■	■	■	■
D2	■■■	■	■	■	■	■	■
H1	■■■	■■	■	■	■	■	■
H2	■■■	■■■	■	■■	■	■	■
H3	■■■	■■	■	■	■■■	■	■
H4	■■■	■■	■	■	■■■	■	■
H5	■■■	■	■	■	■	■	■
H6		■■■	■■	■	■	■	■
H7		■■■	■■	■	■	■	■
H8	■■	■■■	■	■■	■	■	■
W1		■■■	■	■■■	■	■	■
W2		■■■	■	■■	■	■	■
W3		■■■	■■	■■	■	■	■
W4		■■■	■	■■■	■	■	■
W5		■■■	■	■■■	■	■	■
W6		■■■	■	■■■	■	■	■
Z1	■■	■■■	■	■■■	■	■	■
Z2		■■■	■	■■■	■	■	■
Z3		■■■	■	■■	■	■	■
Z4	■■	■■■	■	■■	■	■	■
Z5	■■	■■■	■	■■	■	■	■

5.3.1 Digitizing

After some years, use of the ecoregions and ecodistricts called for a digitized version (Klijn & Van 't Zelfde, 1992) with enhanced spatial accuracy in order to allow automatic overlay with other geographical data and various subsequent analyses.

The spatial accuracy of the original ecoregion and ecodistrict maps was determined by the basic maps used for hand-drawing these on polyester film. These maps had an accuracy of between 1: 2,000,000 and 1: 250,000.
However, the decision to digitize the ecoregions and ecodistricts as polygons (vector format) in ARC/INFO required higher accuracy standards to allow for overlays to be made with geographically more detailed data, aggregation of the results of predictive modelling, and attribution of data from monitoring programmes. This necessitated the use of basic data from 1: 250,000 to 1: 100,000 maps, *i.e.* about a factor 4 to 10 more accurate. More accurate digitizing is not recommended for two reasons. Firstly, an accuracy may be suggested which is not in agreement with the concept of ecoregions and ecodistricts. After all, one does not want to encourage local authorities to determine in which ecodistrict a certain location is situated, especially when national or regional management measures would be designed for individual ecodistricts. Secondly, the use of very large scale maps for digitizing may influence the fluency of boundaries between ecoregions and ecodistricts to such an extent that extreme irregularity results, requiring subsequent smoothing.

The following basic maps were used for the digitizing:

- Soil Map of the Netherlands 1: 250,000 (Steur *et al.*, 1985);
- Topographical Map 1: 100,000 (copy of the 1: 50,000 topographical map; National Topographical Service);
- Geological Map of the Netherlands 1: 600,000 (Van Staalduinen, 1979);
- Polyester film (1: 250,000) with the most recent findings concerning the precise location of the fault lines of the '*Centrale Slenk*' (a graben, see Strahler 1975, page 496) made available by the National Geological Survey.

It may be noted that the Geological Map does not meet the requirement of a scale larger than 1: 250,000. However, no more accurate data were available to define the very few boundaries derived from this map.

Different maps were used for digitizing different boundaries, depending on the mapping characteristics used to distinguish the various ecoregions and ecodistricts. This necessitates a further specification of the basic data used for digitizing the boundaries between ecoregions and ecodistricts. Since the boundaries of the ecoregions correspond with those between ecodistricts, only the latter are specified in Table 5.3.

The digitizing was carried out on a digitizing tablet, allowing for generalization and smoothing while digitizing. As ecoregions and ecodistricts have a hierarchical relation to each other, one digitizing procedure sufficed (Figure 6.1). The attribute file has been set up in such a way that it specifies both the ecoregion type and the ecodistrict type.

Table 5.3 Basic maps used for digitizing the boundaries of each ecodistrict type.

L1	Geological map/ soil map
L2	Soil map
P1	Soil map
P2	Soil map
P3	Soil map
P4	Geological map/ soil map
P5	Soil map
P6	Soil map
P7	Soil map
P8	Soil map
P9	Soil map
P10	Soil map
P11	Soil map
P12	Soil map
P13	Soil map
P14	Polyester film, RGD
D1	Soil map, map of the coastal dunes (1: 100.000) by Bakker *et al.* (1981)
D2	Soil map, map of the coastal dunes (1: 100.000) by Bakker *et al.* (1981)
H1	Soil map
H2	Soil map
H3	Soil map, Topographical map
H4	Soil map, Topographical map
H5	Soil map, Topographical map
H6	Topographical map
H7	Topographical map
H8	Topographical map
W1	Topographical map
W2	Topographical map
W3	Topographical map
W4	Topographical map
W5	Topographical map
W6	Topographical map
Z1	Arbitrary, border of the Netherlands' land mass
Z2	Topographical map
Z3	Topographical map
Z4	Arbitrary, border of the Netherlands' land mass
Z5	Arbitrary, border of the Netherlands' land mass

5.3.2 Accuracy and applicability

Any policy-making process requires use of information of adequate detail. In the case of classifications and maps this means that they should not be used for planning or managing land units smaller than, or even the same size as, the smallest possible unit on the map. In this context, the concepts of basic mapping unit (b.m.u.) and basic planning unit (b.p.u.) as put forward by Vink (1963; 1975, page 95) are relevant. The *basic mapping unit* is the smallest unit that can be drawn on a map corresponding to its real area and shape. It is a function of the mapping scale, because it is determined by technical

constraints, such as drawing with a pen, or digitizing with a mouse. The size of the smallest mapping unit is about 0.25 cm^2 (1/2 x 1/2 cm^2 or 1 x 1/4 cm^2). The *basic planning unit* is defined as the smallest unit for which sound policy or management decisions can be made on the basis of a given map. The relationship between basic mapping unit and basic planning unit can be defined as a factor. This permits the basic planning unit to be derived from the basic mapping unit, and *vice versa*. For some frequently used mapping scales, the basic mapping unit, the factor, and the resulting basic planning unit are specified in Table 5.5.

Table 5.5 The relationship between basic mapping unit and basic planning unit (after: Vink, 1963; Van de Weg, 1981).

MAPPING SCALE	BASIC MAPPING UNIT		FACTOR	BASIC PLANNING UNIT	
1: 10,000	0.0025	km^2	2	0.005	km^2
1: 50,000	0.06	km^2	4	0.25	km^2
1: 100,000	0.25	km^2	4	1	km^2
1: 250,000	1.6	km^2	6	9.6	km^2
1: 1,000,000	25	km^2	8	200	km^2

The basic data used for digitizing determine the mapping scale that should be used for output purposes, *i.e.* paper maps. It is estimated that this should be no more detailed than 1: 500,000 for the ecodistrict map. This corresponds with a basic mapping unit of about 6 km^2, yielding a basic planning unit of about 50 km^2. For applications such as overlay, correspondence analysis or modelling the accuracy of the boundaries in comparison with ground-truth is important. The accuracy of our digitization is estimated to be within 0.5 mm of the boundaries on the maps used for this purpose. Although these were mainly at scales of 1: 250,000 and 1: 100,000, it is assumed that slight deviations from ground-truth in the base maps do not allow us to guarantee a higher accuracy than within 250 m in the field. This corresponds with an accuracy of 0.5 mm from the ground-truth at an output scale of 1: 500,000.

5.4 Discussion

In this chapter we have tried the proposed guidelines concerning the selection of classification characteristics by elaborating classifications of ecoregions and ecodistricts. These were primarily set up as a mapping exercise aimed at 'ecoregionalization', to improve communication about environmental quality and environmental hazards between scientists and policy-makers.

It proved difficult to design a classification intended primarily to serve as a map legend which is also consistently systematic, as we have to comply with practical constraints imposed by the mapping scale and the nature of the subject matter, *viz.* ecoregions and

ecodistricts. We had to sail between the Scylla of ecological theory and the Charybdis of pragmatism to come up with a comprehensible map and legend. This implied a relatively 'holistic' procedure with a comparatively loose application of our own guidelines. Consequently, the classification lacks evident systematics, which is understandable for two reasons.

Firstly, according to the guidelines, the classifications are based largely on geomorphological characterictics. The mainstream in geomorphological classification and mapping is genetically based and relatively qualitative in its definitions, whereas quantitative approaches seem to have less ecological relevance because of a lack of predictive power for dependent ecosystem components. For example crest, footslope, or pointbar are more informative then absolute altitude or slope angle. As has been stated earlier, a qualitative emphasis also enhances communication, because it relates to what can be understood with common geographical (secondary school) knowledge.

Secondly, the number of map units on the ecoregion and ecodistrict maps of the Netherlands are but few. Therefore, there is less need for a systematic classification, which is intended primarily for 'typification' (Zonneveld, 1994), as each mapping unit can be shown on the map and is relatively unique. This may be very different when ecodistricts are to be classified for larger areas, requiring international harmonization.

In this chapter it has also been demonstrated that the guideline as to which ecosystem components should be used to retrieve classification characteristics from (Figure 5.1) may require flexible application. Tables 5.2 and 5.3 reveal that each classification boundary between ecoregions and/or ecodistricts is based on characteristics from several ecosystem components in varying degrees of importance, although remaining within the range indicated in the guideline. It appears that the framework for classification should indeed be interpreted as a guideline, and certainly not as a straitjacket.

As for the applicability of the ecoregion and ecodistrict maps of the Netherlands, we consider their actual use since initial compilation in 1988 for both analysis and presentation a sign that the primary aim of the ecological regionalizations has been achieved. Among the main applications are susceptibility assessment (Klijn 1991; see also Chapter 7) and quality assessment (Klijn *et al.*, 1990; Nip *et al.*, 1992; Latour & Groen, 1995). The maps are also used for geo-referencing and for generalizing the results of sophisticated predictive modelling with detailed geographical data in order to create maps concerning recognizable and well-known geographical units which enhance communication with policy-makers. Finally, ecodistricts are regarded as a good basis for regional environmental policy aimed at simultaneously tackling all environmental problems within a specific area (Klijn & Laansma, 1990; Ministry of VROM, 1990).

The usefulness of this approach to ecological regionalization is now recognized elsewhere in Europe as well, particularly in Flanders (Antrop *et al.*, 1993) and Germany, primarily because of its advantages for 'state-of-the-environment' reporting.

6 Evaluating the ecoregion and ecodistrict maps[1]

6.1 Introduction

The classifications of ecoregions and ecodistricts of the Netherlands were the result of deductive, top-down reasoning. The integration of knowledge from all the basic maps was carried out primarily in the mind of the mapper, who could do no better than to explain as explicitly as possible what lines of thought he followed. Also, as the lists of predominantly used classification characteristics and mapping characteristics (Tables 5.1 and 5.2) demonstrate, each boundary is based on a specific set of classification and mapping characteristics, which may differ from the set used for another boundary (see also Table 5.3 for the component maps we used for the digitizing). This is the inevitable consequence of an *a priori* integrated approach (Zonneveld, 1979; 1994). However, the apparent subjectiveness of this deductive and integrated approach also provokes substantial criticism for it being arbitrary. This may be accounted for by regarding the results of such an approach as a set of hypotheses, thus making it subject to testing.

As for the ecoregions and ecodistricts, we might test many aspects: for example, whether each distinction is equally relevant, or whether the 'mapper' consistently used the mapping characteristics he claims he used, or we could establish the ecological relevance of the maps by comparing them with the distribution patterns of biota. Here we limit ourselves to tests concerning the consistent use of mapping characteristics, for which we had data available.

6.2 Testing material

We used data from a geographical database that is being established in the context of the Landscape Ecological Mapping of the Netherlands (LKN; Canters *et al.*, 1991; Bolsius *et al.*, 1994), which contains ecologically relevant basic data on abiotic and biotic components. These data originate either from independent mappings, mainly at a scale of 1: 50.000 and drawn up over decades of standardized mapping of soil, geomorphology, etc., or from floristic and faunistic inventories by both governmental and non-governmental organizations. Depending on the component, these indicate percentage area or mere presence in 1 km^2 grid cells. None of the data included in LKN were used directly for the preliminary mapping or for the digitizing of ecoregions and ecodistricts.

All data on abiotic components in LKN were regarded as potentially relevant and sufficiently independent for testing the consistency of the digitized ecoregion and ecodistrict maps. We used information on geomorphology, (vertical) groundwater movements, groundwater table classes and soils, because these should correspond sufficiently

[1]This chapter and the former are slightly revised versions of halves of Klijn, F., R.W. de Waal & J.H. Oude Voshaar, 1995. Ecoregions and ecodistricts: ecological regionalizations for the Netherlands' environmental policy. *Environmental Management* 19/6: 797-813

well with the classification characteristics to allow for their use as mapping characteristics. Moreover, from an ecological point of view these abiotic components can be regarded as still relatively independent (*sensu* Jenny, 1941; Kwakernaak, 1982).
The data on biotic components might have been used for testing the ecological relevance of the maps, because they can be regarded as dependent components. However, the biotic data suffer from poor coverage, blurring the picture. In addition, the distribution of biota is heavily influenced by human disturbances. For evaluating the *maps* of ecoregions and ecodistricts, the biotic data are regarded as insufficient. Therefore, we have used only the aforementioned abiotic data.

6.3 Method

Evaluating the use of mapping characteristics for the maps of ecoregions and ecodistricts requires some kind of multi-variate analysis (Mardia *et al.*, 1979; Krzanowski, 1988). We decided to use canonical variate analysis (CVA), because this is the most suitable for explaining groups, *i.e.* ecoregion types and ecodistrict types. Canonical variate analysis is based on maximizing the ratio of between-group variability to within-group variability in each direction (Digby *et al.*, 1989).
We used the 5th release of GENSTAT (Digby *et al.*, 1989) for our calculations, with a maximum capacity of 40,000 units (at a later stage the capacity was increased). This maximum capacity set limits on the number of grid cells in relation to the number of variables. The LKN database contains many records on about 36,000 terrestrial grid cells.
The programme yields:

- CVA-values which indicate the discriminating value of each variable or combination of variables. The higher the value, the larger its discriminating value;
- the number of 'misfits', *i.e.* cells that would be classified differently if solely the respective variable or combination of variables were used consistently;
- a table with the frequency distribution of the number of grid cells for each variable within the evaluated groups, *i.e.* ecoregion types or ecodistrict types;
- a reclassification of the misfits;
- a table with the frequency distribution of the number of grid cells for each variable after reclassification.

As the LKN database contains data on 1 km^2 grid cells, we first transformed the polygon maps of ecoregions and ecodistricts (figure 6.1) into grid maps. This transformation was achieved by overlaying the maps with the grid and attributing all boundary cells to the unit with the largest share of a cell. Because the LKN data on geomorphology, vertical groundwater movement, groundwater level classes and soils are mainly qualitative (or nominal) characteristics, each classification unit, *i.e.* soil type, etc., had to be regarded as variable. For each variable, its surface area within a 1 km^2 grid cell is specified as a percentage. In this way, we would have had to cope with about 250 geomorphological variables, 10 groundwater movement variables, about 200 soil variables and 8 groundwater table variables. This proved too many in relation to GENSTAT's (then) maximum capacity.

Figure 6.1 Digitized version of the ecodistrict map in polygon format.

Therefore, we first generalized these component classifications by discarding all distinctions that were supposed not to have any ecological significance and consequently no discriminant value. Thus we ended up with:

- geomorphology 42 variables
- groundwater movement 6 variables
- groundwater level 7 variables
- soil 36 variables

Subsequently, we again reduced the number of data by taking into account only the dominants.

After these reductions of data numbers GENSTAT could cope with a maximum of about 3,500 grid cells in one run. We give an illustration for a sample area of 2,250 cells between The Hague and Amersfoort[2]. It stretches from the middle of the country to the coast, in an attempt to cover as many ecoregion and ecodistrict types as possible.

We first evaluated the distinctions between ecoregions and subsequently the subdivision of ecoregions into ecodistricts using the ecoregions of different type as stratified samples. The analysis we carried out can best be described by the following questions:

- what individual component is the most discriminating?
- what additional component then yields the greatest improvement?[3]
- and, finally, what is gained by adding a third and fourth component?

6.4 Results

For ecoregions, Figure 6.2 gives the CVA-values for the distinctions between ecoregions within the sample area. Of the single components soil appears to be the most discriminant, having by far the highest CVA-value. Combination with geomorphology, however, yields a substantial improvement, which can be increased only slightly by also adding the groundwater level class. This latter improvement is not very significant, however.

Based on the combination of soil and geomorphology, we reclassified the grid cells. Plate 6.1 shows the original map (A), the map resulting from automated reclassification (B) and a map of the cells that were classified differently: the 'misfits' (C). The misfits comprise almost 2% of the sample area in 1 to 4 km^2 patches. Most differences occur along the borders between the ecoregions, but some outliers also show up.

[2]At the time the analyses were carried out, soil and groundwater level data covered about 85 % of the country, groundwater movements covered the whole country, but geomorphology only 40 %.

[3]In fact, it would be preferable to also evaluate combinations of individually weakly discriminating components, as combinations of such components might be very discriminating. This would have required either a multitude of analyses or an adaptation of the computer programmes. We estimated that it would not significantly influence the results.

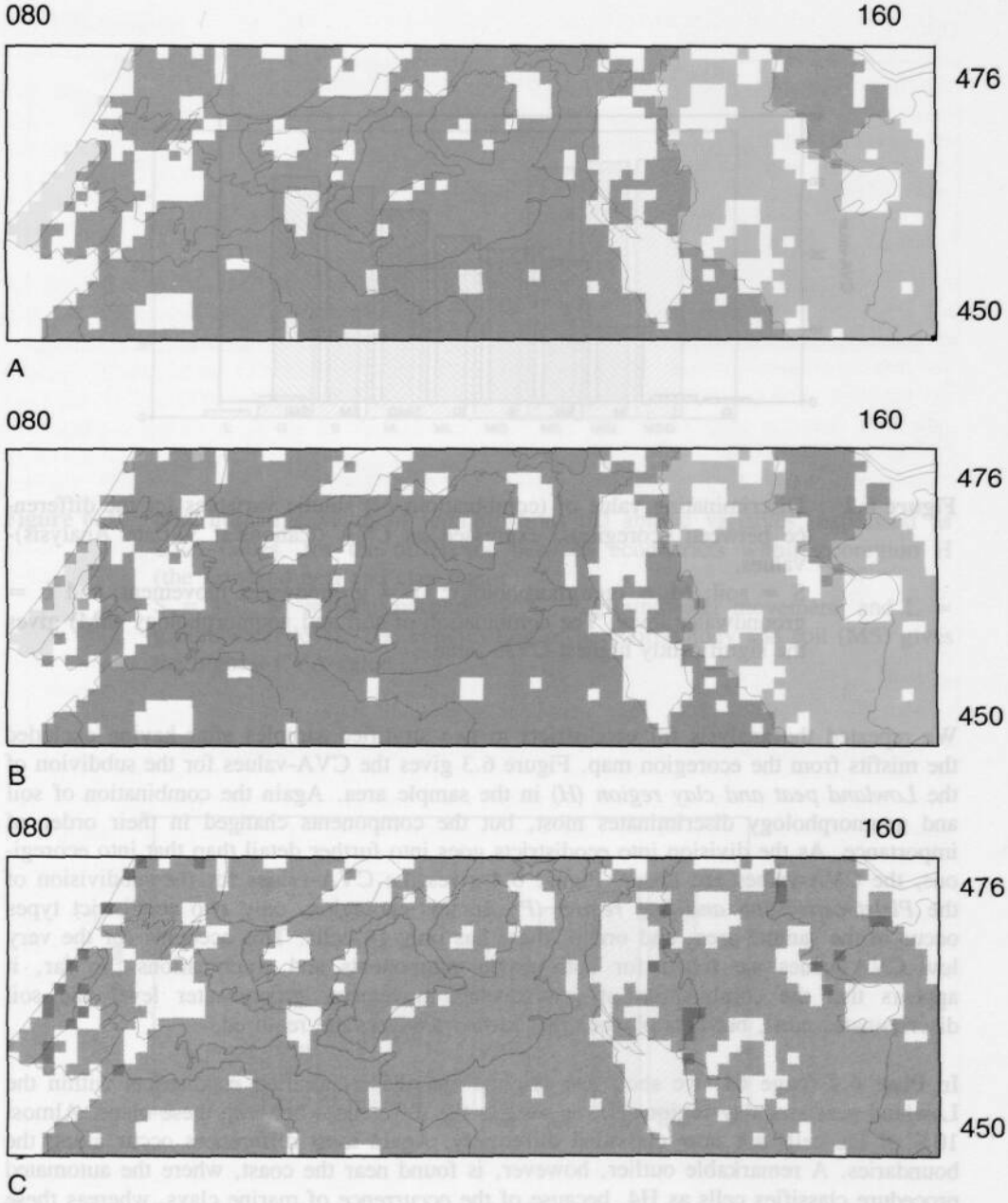

Plate 6.1 The pattern of ecoregions in the sample area as originally mapped and digitized (Klijn 1988; Klijn and Van 't Zelfde, 1992) but converted to grid cells (**A**), from automatic reclassification on the basis of soil and geomorphology (**B**), and the cells that are classified differently (**C**).

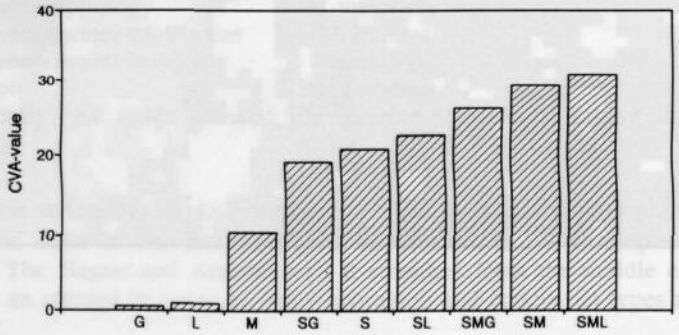

Figure 6.2 Discriminating value of (combinations of) abiotic variables for the difference between ecoregions, expressed as CVA (Canonical Variate Analysis)-values.
S = soil; M = (geo)morphology; G = groundwater movement; and L = groundwater level. The combination of soil and geomorphology (SM) gives the significantly highest CVA-value.

We repeated the analysis for ecodistricts in two stratified samples after having excluded the misfits from the ecoregion map. Figure 6.3 gives the CVA-values for the subdivion of the *Lowland peat and clay region (H)* in the sample area. Again the combination of soil and geomorphology discriminates most, but the components changed in their order of importance. As the division into ecodistricts goes into further detail than that into ecoregions, the CVA-values are lower. Figure 6.4 gives the CVA-values for the subdivision of the *Pleistocene sand and bog region (P)*. In this ecoregion, only two ecodistrict types occur in the sample area, and one of these has only 14 cells. This accounts for the very low CVA-values we found for both single components and combinations. So far, it appears that the combination of groundwater movement, groundwater level and soil discriminates most, but is is obvious that further analyses are required.

In Plate 6.2 (page 64) we show the original and the reclassified ecodistricts within the Lowland peat and clay region (H), as well as the differences between these maps. Almost 10% of the cells are now classified differently. Again most differences occur along the boundaries. A remarkable outlier, however, is found near the coast, where the automated procedure classifies cells as H4, because of the occurrence of marine clays, whereas these small intrusions were taken together with the fluvial H2 on the original map.

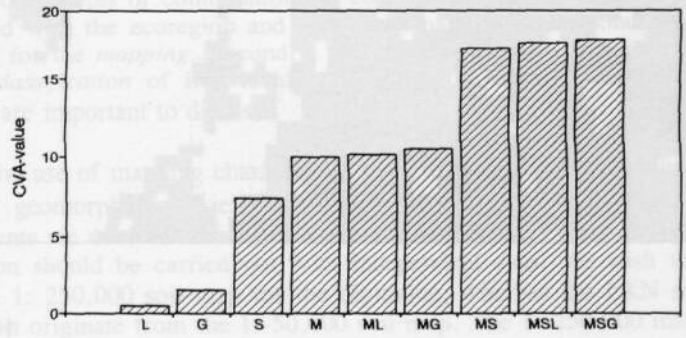

Figure 6.3 Discriminating value of (combinations of) abiotic variables, expressed as CVA-values, for the difference between ecodistricts within ecoregion H (the Lowland peat and clay region).
S = soil; M = (geo)morphology; G = groundwater movement; and L = groundwater level. The combination of geomorphology and soil (MS) gives the highest CVA-value.

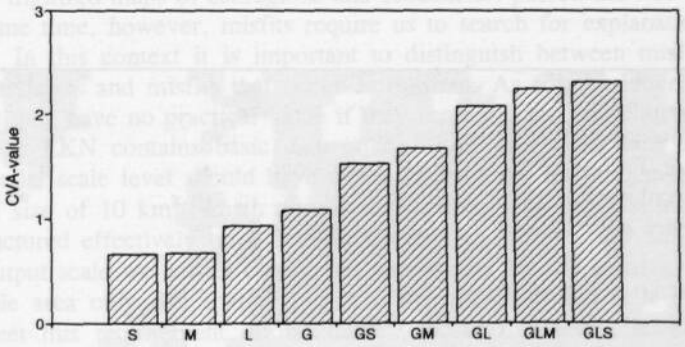

Figure 6.4 Discriminating value of (combinations of) abiotic variables, expressed as CVA-values, for the difference between ecodistricts within ecoregion P (the Pleistocene sand and bog region). S = soil; M = (geo)morphology; G = groundwater movement; and L = groundwater level. The combination of groundwater movement and groundwater level (GL) gives the significantly highest CVA-value.

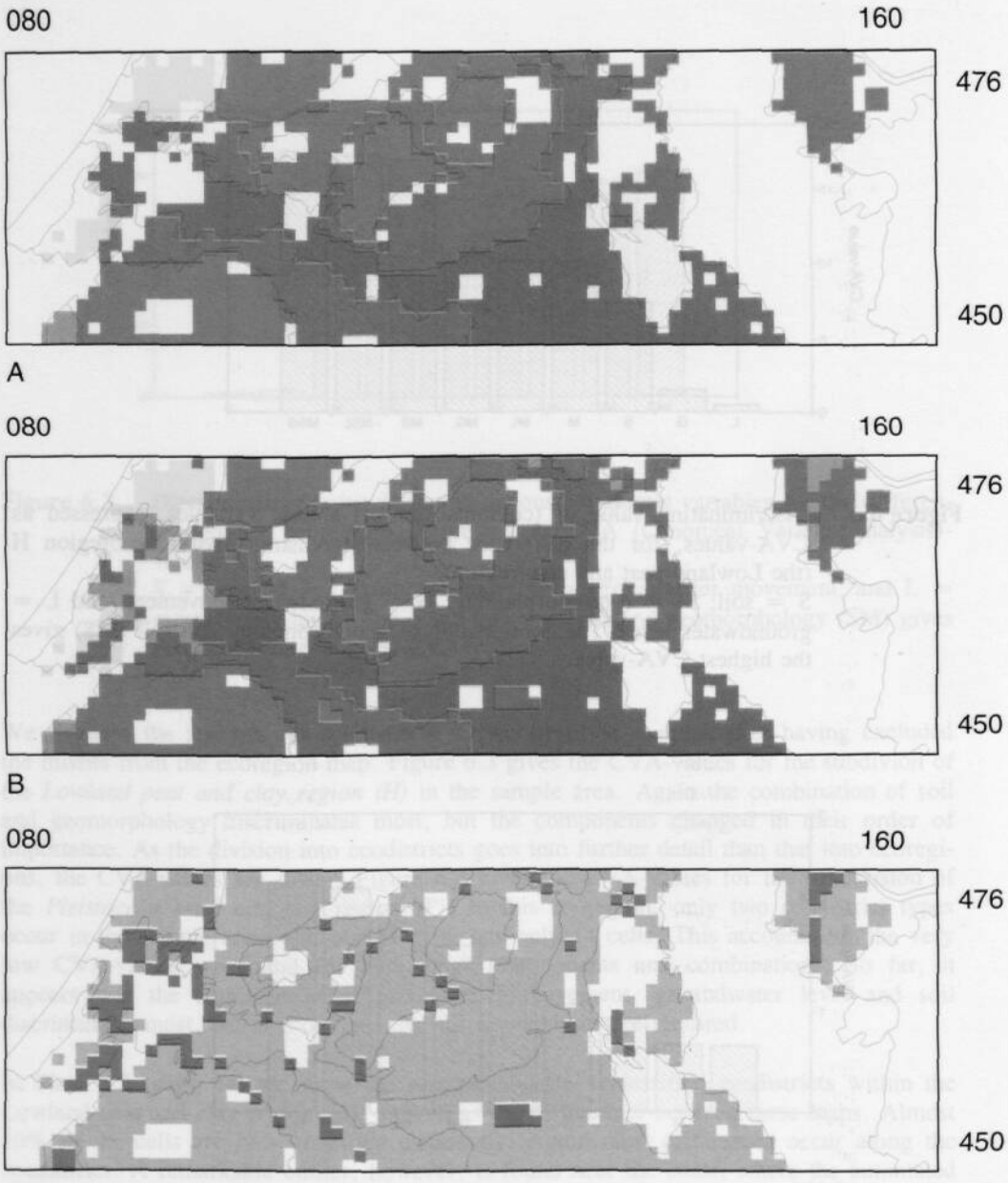

Plate 6.2 The pattern of ecodistricts within the stratified sample concerning the Lowland peat and clay region, as originally mapped and digitized (Klijn 1988; Klijn and Van 't Zelfde, 1992) but converted to grid cells (**A**), from automatic reclassification on the basis of geomorphology and soil (**B**), and the cells that are classified differently (**C**).

6.5 Discussion

The above evaluation yielded two important results. Firstly, it demonstrated that the abiotic components or combinations of components which were found to be most closely connected with the ecoregion and ecodistrict patterns correspond with the characteristics we used for the *mapping*. Secondly, the automatic reclassification revealed discrepancies in the *classification* of individual cells. Both the number of these 'misfits' and their location are important to discuss.

As for the use of mapping characteristics, the evaluation revealed that the combination of soil and geomorphology yielded the highest correspondence. These were precisely the components we used for the digitizing (*i.e.* the mapping). As for the requirement that an evaluation should be carried out with independent data, we wish to emphasize that we used the 1: 250,000 soil map for the digitizing, whereas the LKN soil data used for the evaluation originate from the 1: 50,000 soil map. The 1: 250,000 map is based on fewer, different and partly older inventories. As for geomorphology, we often used the 1: 100,000 topographical map for the digitizing, which permitted the derivation of quite a number of ecologically relevant geomorphological boundaries, whereas the LKN data on geomorphology used for the evaluation originate primarily from the 1: 50,000 geomorphology mapping (*cf.* Maas *et al.*, 1995, page 22).

The discrepancies between the original maps and the results of automatic reclassification were numerous. However, the percentage of misfits amounted to slightly less than 2% in the case of ecoregions (Plate 6.1 C) and just below 10% in the case of the ecodistricts of the Lowland peat and clay region (Plate 6.2 C). Taking into account that an accuracy of over 70% is generally regarded sufficient for soil maps, we may consider the number of misfits highly acceptable. In general, therefore, we feel justified in stating that the originally digitized maps of ecoregions and ecodistricts passed the test.
At the same time, however, misfits require us to search for explanations and to consider revisions. In this context it is important to distinguish between misfits that are located along boundaries and misfits that occur as outliers. As for the latter, we should bear in mind that they have no practical value if they represent too small an area for the scale of interest. As LKN contains basic data on a 1 km^2 grid basis, any regionalization at a higher spatial scale level should have a size at least one magnitude larger. This means a minimum size of 10 km^2, which almost corresponds with the smallest mapping unit that can be pictured effectively on a 1: 500,000 map (6.25 km^2, see Table 5.5). This is the largest output scale we earlier considered appropriate for the digitized ecodistrict map. In the sample area only one spatial unit near the coast (around cells 90/465 in Plate 6.2) might meet this requirement of minimum size. In this area, the automated procedure classifies many cells as H4, because of the occurrence of marine clays, whereas these small intrusions were taken together with the fluvial H2 on the original map. It might exceed the size of the smallest mapping unit, but the picture is blurred because, firstly, it concerns a very gradual transition between fluvial and marine clays at the mouth of the River Rhine causing a mixture of H2 and H4, and, secondly, the area is largely urbanized and consequently has become discontinuous. The rationale for regarding it as part of the fluvial H2 in the original ecodistrict map was the judgement that for such a small area the genesis of a clay soil is ecologically insufficiently relevant in comparison to its actual

properties to justify its distinction.
When we discard all too small outliers, the number of misfits decreases substantially again.

With respect to an evaluation of maps by regarding them as sets of hypotheses as described above, we wish to emphasize that any such evaluation depends strongly on the appropriateness of the sample data used for this purpose. The size of the sample, the way it is stratified and the characteristics taken into account are all known to affect the outcome.

As for our evaluation with data from LKN, we were lucky to have full coverage for the components taken into account for the evaluation, implying that the size of the sample is unquestioned. In contrast, LKN does not cover the whole set of possibly relevant abiotic components, excluding, for example, geological formations and their composition and groundwater at greater depth. In this context, we recall that the classification and mapping of ecoregions and ecodistricts was based on many more maps and data than used for the evaluation. In other words, the original classification and mapping are of a much more integrated nature (*cf.* Zonneveld, 1994) than can in practice ever be achieved by *a posteriori* integration. In addition, the LKN data are classified data, which means that at an earlier stage they were ordered and generalized in an arbitrary and irreparable manner. Both the degree of detail and the principles of generalization may influence the outcome. Finally, GENSTAT's capacity compelled us to reduce the number of variables, for which reason we only took into account the dominants, perhaps at the expense of two smaller classes from the same 'family'.
Because of the aforementioned considerations, we consider the results of the automated reclassification not necessarily better than the original. Rather, we regard the differences between the maps as an aid on the road to further improvements[4].

[4]Meanwhile, the evaluation of ecoregions and ecodistricts has been extended to the whole country, which became possible after enlarging GENSTAT's capacity (*cf.* De Waal, 1996).

7 Susceptibility assessment of ecodistricts[1]

7.1 Linking processes to patterns

In Chapter two we stated that ecosystem classifications are supposed to be tools for environmental policy analyses. After our evaluation of the ecodistrict map, we should therefore try its applicability.

In this chapter, we shall focus on the susceptibility assessment of ecodistricts for some environmental hazards in order to exemplify the procedure which we classified as a variation on the theme of physical land evaluation in Section 2.3.1. This requires an adequate linking of the processes occurring in ecosystems as a result of human activities, or through disturbance by human influence, to structural characteristics of the legend units which can be used for land evaluation. It shall be demonstrated that, in this context, the recognition that ecological land units can be regarded as correlative complexes is essential. Otherwise, it would be impossible to (semi-)quantify the parameters that control the ecosystem response for each ecodistrict type because of a lack of geographical data. Consequently, the assessment of ecodistrict susceptibility would also be impossible.
Before exemplifying the procedure of susceptibility assessment, we shall first establish the relationship between environmental 'themes' on the one hand and the hierarchy of ecosystem components and spatial scale levels on the other in a very general and simplified way, because this helps to focus on the most important ecosystem components as well as on the most appropriate spatial scale level(s) for each theme.

7.2 Environmental themes as chains of processes in ecosystems

An analytical approach to any environmental problem requires that we first exactly define the environmental *hazard*[2] in relation to the societal function or value of the environment at risk. Only then can we concentrate on the related relevant processes and the related ecosystem characteristics. This apparently conflicts with a desire to develop multi-purpose tools, as, unfortunately, an almost infinite number of environmental hazards can be distinguished.

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[1]This chapter is based partly on texts from Klijn, F., 1991. Environmental susceptibility to chemicals: from processes to patterns, with special reference to mapping characteristics and spatial scales. In Batjes, N.H. & E.M. Bridges (eds.), 1991. *Soil vulnerability to pollution in Europe.* ISRIC, Wageningen. pp. 9-22 and Klijn, F., 1994. Spatially nested ecosystems: guidelines for classification from a hierarchical perspective. Pages 85-116 in Klijn, F. (ed.), 1994. *Ecosystem classification for environmental management.* Kluwer Academic Publishers, Dordrecht.

[2]We use the term hazard to indicate an environmental danger, that is a potential problem which may become actual only when societal functions and/or values are at stake; hence, it relates to susceptibility (see Section 2.3.1). A *risk*, in contrast, relates to vulnerability, which implies effects on societal functions and/or values.

To structurize the multitude of environmental hazards, however, we can group them into a small number of central *themes* as distinguished for the Netherlands' environmental management (RIVM, 1989; see also Section 1.1). These themes are relatively exclusive groups of related environmental hazards. For reasons of convenience we shall follow this distinction, emphasizing that the list is not limitative and that some themes partly overlap:

- climatic change
- acidification
- pollution (with toxic substances; also called 'dispersion' or intoxication)
- eutrophication
- 'desiccation' or 'drought damage'
- destruction
- fragmentation
- disturbance (by noise or odour)

The majority of these themes can be understood as chains of ecologically relevant processes which subsequently affect the structural characteristics of abiotic and biotic ecosystem components. In terms of the system levels distinguished by Chorley & Kennedy (1971; see Section 3.2) they could well be studied as process-response systems that are arranged in cascades. These chains of processes follow the direction of fluxes of energy and matter, as indicated in the hierarchical model (Figure 4.1) by the downward arrow. Most environmental problems result from such fluxes of energy and matter and relative dependence among ecosystem components. In all cases, they may finally result in harm to flora and/ or fauna, including human beings.

When wishing to zoom in on a specific environmental hazard, our analysis may be guided by the point(s) of impact in terms of the prime ecosystem component(s) which are initially and truly affected. We specify the main points of impact with the resulting cascades of effects in relation to our hierarchy of ecosystem components in Figure 7.1, emphasizing that it is, of course, a gross generalization.
However, it is intended to illustrate that the point of impact of environmental hazards differs widely, from climatic change and acidification, which primarily influence atmospheric composition, to the destruction of vegetation or fauna due to physical interventions such as land use change (Klijn, 1988; RIVM, 1989). This notion is important, because, owing to these different points of impact, the processes act on different time scales and the effects become manifest after longer or shorter periods of time. Usually, environmental hazards affecting the atmosphere at large result in changes of vegetation or fauna only after long periods, as is exemplified by acidification: the effects of sulphuric acid on the environment were already observed and drawn attention to in the seventeenth century (Evelyn, 1661), but it was not until the seventies of this century that massive societal awareness really arose. In contrast, the destruction of the vegetation by clear-cutting or the disturbance of the fauna by noise generally have immediate and very obvious effects.

The relationship between the point of impact and the hierarchy of ecosystem components may also help to decide on the most relevant spatial scale level to deal with a problem[3]. This follows from the fact that, on the one hand, the determinant processes are connected to the ecosystem components according to Figure 7.1, whereas, on the other hand, we have connected the ecosystem components with classification levels for different spatial scales (Figure 4.3).
Now, obviously, analyses at spatial scale levels above the highest point of impact seem quite useless. In contrast, a scale level on or just below the point of impact may be the most appropriate to focus on, especially when the most important conditioning processes are controlled by the corresponding component, because this explains the most significant differences in response.

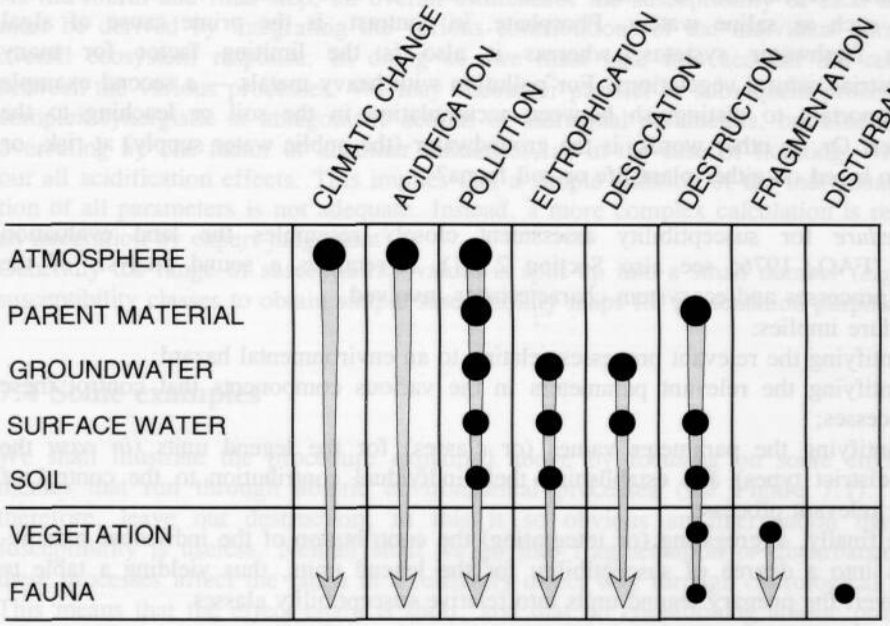

Figure 7.1 Main 'points of impact' of environmental hazards, indicated by dots, and subsequent effect cascades, indicated by arrows. Some environmental themes may impact on a number of different components.

[3]Of course, the extent of an intervention is also important for the decision as to the most relevant spatial scale. This depends especially on the area in which a human activity is carried out, or the number and spatial distribution of emission points.

7.3 General procedure of susceptibility analysis

A proper analysis of any environmental hazard first requires an exact *definition* of the environmental processes concerned, as well as of the societal function or value of the affected environmental components.
For susceptibility assessments in general, it is of utmost importance to first define precisely the environmental processes concerned, and possibly even the detrimental substance(s), as well as the threatened component(s) or ecosystem characteristic(s). The latter is important because different components or characteristics are relevant for different societal functions or represent different environmental values. Only through clear specification does a subsequent normative assessment become possible, for example when choices have to be made in relation to different preferences for land use.
For the environmental theme 'eutrophication', for example, we must distinguish between nitrate and phosphate problems, because they behave differently. In addition, nitrate is primarily a risk factor for public water supply from groundwater, though it may also affect natural vegetations or cause algal blooms in ecosystems without phosphorus limitation, such as saline waters. Phosphate, in contrast, is the prime cause of algal growth in freshwater systems, whereas it also is the limiting factor for many (semi)terrestrial natural vegetations. For pollution with heavy metals — a second example — it is important to distinguish between accumulation in the soil or leaching to the groundwater. Or, in other words: is the groundwater (the public water supply) at risk, or a foodchain based on either plant life or soil fauna?

The *procedure* for susceptibility assessment closely resembles the land evaluation procedure (FAO, 1976; see also Section 2.3.1). It requires a sound analysis of the ecological processes and ecosystem characteristics involved.
The procedure implies:

- identifying the relevant processes relating to an environmental hazard;
- identifying the relevant parameters in the various components that control these processes;
- quantifying the parameter values (or classes) for the legend units (*in casu* the ecodistrict types) and establishing their individual contribution to the control of any relevant process;
- and finally, aggregating (or integrating) the contribution of the individual parameters into a degree of susceptibility for the legend units, thus yielding a table to convert the primary legend units into relative susceptibility classes.

So, as the first step, an environmental hazard must be unraveled into a number of relevant processes. This may include counteracting processes, such as the buffering of acidification by the solubilization of $CaCO_3$ or by flooding, or, alternatively, triggered processes, such as the mineralization of organic matter or a pH decline, both resulting from groundwater lowering. Such an unraveling into individual processes which primarily determine ecosystem response yields a list of processes to be taken into account that is specific for each environmental hazard.

As the second step, we must identify the structural ecosystem characteristics that control these processes. These conditioning factors will subsequently be called *parameters*,

because they may function as parameters in mathematical prediction models for ecosystem-specific effect predictions (Sinnige *et al.*, 1994; Traas *et al.*, 1995).

As the third step, we must quantify the various parameter values for the different legend units (*in casu* ecodistrict types) and establish the contribution of each parameter to individual control processes. The latter is important for the decision as to what level of detail is needed in quantification.
Quantification of the parameters will generally be accomplished in terms of ranges (or relative classes) instead of real values. Also, many parameters will not have been used as classification characteristics; they may even belong to ecosystem components that have not been used for the classification. Fortunately, the existence of the aforementioned correlative complexes permits a quite accurate estimate of the value and internal spatial variability (range) within the classification units for those parameters that have not been measured separately.

As the fourth and final step, an overall estimate of the susceptibility of each legend unit must be derived by integrating the various contributions of the individual parameters to overall ecosystem response. In doing so, we must take into account the relationships between the various processes. We may encounter parallel or subsequent processes, more complex synergistic or antagonistic actions of individual parameters, but also an absolute overruling by one factor of all other influences, as in the case of flooding, which wipes out all acidification effects. This implies that a simple addition of the individual contribution of all parameters is not adequate. Instead, a more complex calculation is required, or an integration by expert judgement.
Generally the range of susceptibility values is split up into a small number (*e.g.* four) of susceptibility classes to obtain simple susceptibility maps for presentation purposes.

7.4 Some examples

We shall illustrate the procedure explained above by focusing on some environmental themes that run through abiotic environmental processes (see Figure 7.1). We shall, therefore, leave out destruction, as this is so obvious an intervention that defining susceptibility is useless. Neither shall we go into fragmentation or disturbance, because these processes affect the fauna in a relatively direct way through chorological processes. This means that the effect chain is short, and that no conditioning abiotic processes are involved. In addition, they require analysis in greater detail, at ecosection level or even at ecotope level.
For the moment we also leave out climatic change, because this can better be analyzed at less detailed, *i.e.* continental to global, scale levels. Thus acidification, pollution with toxic substances, eutrophication and desiccation remain.

7.4.1 Acidification

Acidification is a natural process, taking place everywhere. However, the rate of acidification has exceeded the rate of counteracting processes such as weathering, erosion,

sedimentation, bioturbation, etc. The most important counteracting process is the buffering in soils. The main buffers are the solubilization of CaCO3, the weathering of silicates, the exchange of cations and the breakdown of (clay) minerals causing the release of first aluminium and finally iron cations into the soil solution (after Ulrich, 1980; Verstraten, 1982; De Vries & Breeuwsma, 1986). The release of aluminium is particularly dangerous, as it may cause deficiency diseases in plants.
In addition to these buffers, which are connected with soil characteristics such as CaCO3-content and clay content, groundwater or surface water may also supply cations in areas subjected to upward seepage or flooding. Also, denitrification can counteract acidification in that it is a proton sink. Denitrification is favoured by anaerobic conditions caused by high groundwater levels and high organic matter contents.

This preliminary analysis serves to identify the set of parameters which control the relevant processes and which are therefore relevant for assessing the susceptibility of a land unit to acidification, resulting in:

RELEVANT PROCESSES	RELEVANT PARAMETERS
$CaCO_3$ buffer	primary $CaCO_3$-content depth of decalcification
silicate weathering	content of weatherable silicates texture (clay and silt content)
cation exchange buffer	texture (clay and silt content) organic matter content
breakdown of clay minerals (Al-buffer) iron(hydr)oxide buffer	texture (clay and silt content) iron(hydr)oxide content
cation supply from elsewhere	origin of surface water flooding cation content of surface water salinity of surface water upward seepage cation content of upward seepage salinity of upward seepage
denitrification	groundwater level (fluctuations) organic matter content (redox)

7.4.2 Pollution with toxic substances: heavy metals and organic micropollutants

Pollution with toxic substances involves many different chemicals with varying behaviour. Some general remarks can be made only on 'great groups' of chemicals, *viz.* the heavy metals and organic micropollutants.

For both heavy metals and organic micropollutants it may be relevant to determine the susceptibility to accumulation in the topsoil as well as to leaching to the deep groundwater. This implies that for both groups of chemicals the direction and rate of seepage are of paramount importance.

Most heavy metals, with the exception of e.g. arsenic, are easily bonded by clay and organic matter, because they behave as cations. Hence, they may also precipitate as carbonate, sulphate, etc. Below, we specify the parameters which are relevant for assessing the susceptibility to either the leaching or the accumulation of heavy metals in the topsoil, in relation to the aforementioned processes:

RELEVANT PROCESSES	RELEVANT PARAMETERS
leaching/ accumulation	seepage (direction and rate)
adsorption	texture (clay and silt content) organic matter content groundwater level (redox)
precipitation	primary $CaCO_3$-content depth of decalcification

In contrast, most organic micropollutants are bonded only by organic matter, but we should immediately call attention to the fact that the chemical properties of these compounds are very diverse and consequently also their behaviour. Differences in degradability are great, too.
For most organic micropollutants the following processes and related parameters seem relevant:

RELEVANT PROCESSES	RELEVANT PARAMETERS
leaching/ accumulation	seepage (direction and rate)
bonding	organic matter content groundwater level (redox)

7.4.3 Eutrophication: phosphate and nitrate

Eutrophication is a problem in the Netherlands, primarily as a result of the enormous manure surpluses, though a considerable nitrate load also originates from households, traffic and industry. The spreading of manure results in the gradual saturation of the topsoil with phosphates with subsequent through-flow to surface waters, whereas nitrate is often leached to the groundwater. The processes and parameters determining the behaviour of phosphate and nitrate are different.

Phosphates are strongly fixed in most soils, especially by Fe- and Al-hydroxides, some clay minerals with Fe or Al, or organic complexes with Fe or Al. In addition, Ca-phosphate may precipitate in CaCO3-rich environments. Phosphate fixation is affected by the redox conditions, and hence by (fluctuations in) the groundwater level.

Nitrate, in contrast to phosphate, is usually very mobile in soils. It may be leached to the groundwater and subsequently be transported downwards and horizontally to zones with upward seepage. However, not all nitrate which is deposited will be leached. Part of it may volatilize upon denitrification under reducing circumstances.

The following parameters are considered the most important for assessing the susceptibility to phosphate saturation and through-flow to surface waters:

RELEVANT PROCESSES	RELEVANT PARAMETERS
adsorption/ fixation	Fe- and Al-hydroxides texture (clay content, esp. Fe and Al) type of organic matter groundwater level (redox)
precipitation	primary $CaCO_3$-content depth of decalcification
dilution	volume of surface water current velocity/tides

For assessing the susceptibility of a land unit to nitrate leaching with subsequent contamination of the groundwater, the following processes and related parameters deserve most attention:

RELEVANT PROCESSES	RELEVANT PARAMETERS
leaching	direction of groundwater flow
denitrification	texture groundwater level (fluctuations) organic matter content (redox)

7.4.4 Desiccation

Desiccation is an overall term to indicate all negative impacts of groundwater lowering (Projectgroep Verdroging, 1989), including indirect effects through declining seepage intensities or the increased spread of surface water over vast surface areas to compensate for water shortages (Fiselier *et al.*, 1992). The most immediate effects of groundwater lowering are a decrease in the water availability for plants in summer and a pronounced influence on aeration and mineralization of soil organic matter in spring, which strongly affects the competition between species in natural vegetations (Runhaar, 1989). In addition, the buffering effect of groundwater may decrease, causing internal soil acidification, which may be even more pronounced when the upward seepage of enriched groundwater diminishes. It is obvious that the most pronounced effects on vegetations can be expected in vegetations that depend on groundwater.
For agriculture and forestry an analysis of desiccation may be based on similar reasoning, while soil subsidence and the drying out of shallow (especially seepage-fed) surface waters may also be attributed to the desiccation problem. However, desiccation is nowadays primarily regarded as a problem for nature conservation (*cf.* Runhaar *et al.*, 1995). A more elaborate analysis of desiccation from a nature conservation point of view is given in Chapter 10, where a prediction model for this environmental theme is discussed. For the moment, we conclude by specifying the aforementioned processes and related parameters for assessing the susceptibility of ecodistricts to desiccation:

DOMINANT PROCESS	RELEVANT PARAMETERS
drying up of sources, brook heads and other small surface waters	presence of sources and small surface waters
subsidence	texture/ parent material organic matter content
decrease of water availability	texture organic matter content groundwater level (fluctuations)
increase of aeration	groundwater level (fluctuations)
mineralization	organic matter content type of organic matter groundwater level
acidification	primary $CaCO_3$-content texture (clay and silt content) groundwater level
change in water chemistry	seepage (direction and rate) cation content of upward seepage salinity of upward seepage origin of surface water cation content of surface water salinity of surface water
cation supply from elsewhere	origin of surface water flooding cation content of surface water salinity of surface water upward seepage cation content of upward seepage salinity of upward seepage

7.4.5 Summary of relevant processes and parameters

From the above, it may be concluded that for the assessment of susceptibility to different environmental hazards different sets of parameters are needed. However, the parameter sets partly overlap, while some parameters may be substituted by alternatives from another set, thus permitting the selection of a limited number of parameters for multi-purpose applications (Table 7.1; *cf.* also Van Duijvenbooden & Breeuwsma, 1987).

With this list of relevant parameters for assessing the susceptibility of ecodistricts to various environmental hazards, we are confronted with the problem that no quantitative data concerning the value and the spatial variability of most parameters are available. This certainly goes for larger geographic areas such as ecodistricts, because both the classification and the mapping characteristics are of a more general nature. However, we may fall back on known correlations between values of many of these parameters and qualitative environmental characteristics which we do know, such as those regarding parent material or soils.

Table 7.1 Conditioning ecosystem characteristics considered the most relevant parameters for assessing the susceptibility to 'acidification', to 'pollution' with heavy metals and/or organic micropollutants (by accumulation and/or leaching), to 'eutrophication' and to 'desiccation'.

ENVIRONMENTAL THEMES / CONDITIONING CHARACTERISTICS	acidification	pollution	eutrophication	desiccation
PARENT MATERIAL				
primary $CaCO_3$-content	●	●	●	●
content of weatherable silicates	●			
texture	●	●	●	●
GROUNDWATER				
groundwater level (fluctuations)	●	●	●	●
seepage (direction and rate)	●	●	●	●
cation content of upward seepage	●	●	●	●
salinity of upward seepage	●	●	●	●
SURFACE WATER				
cation content of surface water	●	●	●	●
salinity of surface water	●	●	●	●
SOIL				
texture (clay content, esp. Fe and Al)	●	●	●	●
depth of decalcification	●	●	●	
organic matter content	●	●	●	●
type of organic matter			●	●
Fe- and Al-(hydr)oxides	●		●	

7.4.6 Quantifying conditioning parameters via correlative complexes

Fortunately, there are firstly certain correlations between characteristics of the dependent ecosystem components and the classification characteristics at the scale level of ecodistricts, *i.e.* parent material, geomorphology and (geo)hydrology. This relies upon the relative dependence of the various components in the model and already allowed us to use characteristics of dependent components as mapping characteristics. Secondly, thanks to the same correlations, the values or value ranges of the relevant parameters correlate rather well with the legend units, *in casu* ecodistrict types. Thus, the recognition of correlative complexes permits a rather accurate estimation of the value and internal spatial variability of parameters which have not been measured separately. Such an estimation may be based on elaborate empirical research, existing databases or expert judgement.

We used the latter method to assess the susceptibility of ecodistricts (Klijn, 1988), but supported by scrutinizing existing maps on single parameters (Van Duijvenbooden & Breeuwsma, 1987).

A number of experts were interviewed and asked for an educated guess about the parameter values of each legend unit. When necessary, interpretations in terms of the discipline of the interviewed person were given in order to enhance communication. Thus, we gradually established semi-quantitative estimates of the majority of parameters.
After some simple calculations, the 'degree of susceptibility' was defined for each ecodistrict type. Next, four susceptibility classes were distinguished and attributed to the mapping units. The resulting preliminary susceptibility maps were discussed with the same experts interviewed earlier in order to test the outcome using their qualitative judgement. The resulting susceptibility maps are shown in Figures 7.2 to 7.9.

7.5 Discussion

Our approach to ecological land classification and mapping did not start from the requirements of the desired applications alone, because this would have implied too much emphasis on processes or parameters which cannot easily be mapped. Instead, we started in a deductive way from a simple hierarchical model of ecosystem components. However, as we demonstrated in Section 7.2, this hierarchical model can be related to environmental hazards quite easily, because these hazards can be depicted as effect chains: they impact on a certain ecosystem component, and subsequently affect the dependent components.
The relationship between environmental hazard and point of impact in terms of ecosystem component on the one hand, and the relationship between ecosystem component and spatial scale on the other, also implies a relationship between environmental hazard and spatial scale: environmental hazards act at different spatial scales. This may be an aid for deciding about the spatial scale for a further analysis of any environmental hazard (*cf.* RIVM, 1989).

At the same time, the examples illustrate that it is indeed possible to match a comprehensive ecological land classification with an analytical zooming-in on processes connected with environmental hazards. This matching is based predominantly on the existence of correlative complexes, which allows for a fair estimation of the parameters needed to assess susceptibility.
Still, the approach is merely semi-quantitative and at times even qualitative. Meanwhile, more sophisticated quantitative approaches have been developed for some of the environmental hazards concerned here (*e.g.* Kuylenstierna & Chadwick, 1991; Downing *et al.*, 1993; De Vries, 1994). These allow for a quantitative underpinning of the susceptibility classes in terms of critical loads. Although these approaches are much more formal and quantitative than our approach, this does not necessarily mean that the outcome is better applicable in practice. The prime reason for this is that, in most cases, some processes are quantified with much more precision, especially acid neutralizing capacity (ANC; De Vries, 1994) but often with the concurrent neglect of other important processes and hence at the cost of width of scope. For example, upward seepage or flooding are usually

neglected, which prohibits nationwide application. Neither do, in our opinion, the apparently more detailed susceptibility maps with a resolution of 1 km^2 published by RIVM (1991) imply a larger applicability. These also lack width of scope, as the role of upward seepage and flooding is not taken into account.
In this context, it is also interesting to notice that the present quantitative approaches heavily rely on either maps of single parameters (Van Duijvenbooden & Breeuwsma, 1987) or parameterizations of existing soil maps (*e.g.* De Vries, 1993). Both are also based on the recognition of correlative complexes, but then applied to individual components, *viz.* geology, geohydrology and soil.

The differences in the presented susceptibility maps support the earlier statement that it is only possible to establish the susceptibility of ecological land units to well-defined environmental hazards, specified by both the relevant processes and the target components. Consequently, it is impossible to speak of 'the' susceptibility of the environment. This implies that it is impossible to compose just one susceptibility map. However, we have shown that one basic map suffices as a geographical basis.
In this context, we also recommend to suppress any desire to integrate a number of susceptibility maps into one map of *'the environment's susceptibility'*, as this causes an undesirable simplification (*cf.* RIVM, 1989, page 367). If the desire to obtain an overview is strong, it is instead recommended to summarize the results of the various susceptibility assessments in a table accompanying the legend to the basic map (*cf.* UNEP/ISRIC, 1990) as exemplified by Table 7.2.

Table 7.2 Overview of the susceptibility of ecodistricts to various environmental hazards. The hazards have been arranged in such a way as to show the similarity of the various ecodistrict types (see Figure 5.3 for a legend of ecodistrict type codes, *i.e.* the first column).

ENVIRONMENTAL HAZARD / ECODISTRICT TYPE	heavy metal accumulation	organic micro-pollutant accumulation	phosphate eutrophication	desiccation	organic micro-pollutant leaching	heavy metal leaching	acidification	nitrate leaching
L1	●●	●		●●	●●	●		●●●
L2	●	●	●	●	●●	●●	●	●●●
P1	●	●	●●		●●	●●	●●●	●●●
P2	●	●	●●	●	●●	●●	●●●	●●●
P3	●●	●●	●●	●●●	●●●	●●●	●	●●
P4	●●	●●	●	●	●●	●●	●	●●
P5	●●	●●	●	●●●	●●	●●	●	●●
P6	●	●	●●	●●	●●	●●	●●	●●
P7	●	●	●	●●	●●	●●	●●	●●●
P8	●	●	●●	●●	●●	●●	●●	●●
P9	●	●●	●●	●●●	●●	●●	●●	●●
P10	●●	●●	●●●	●●●	●●	●●	●	●
P11	●●	●●	●●●	●●●	●●	●●	●	●
P12	●●●	●●●	●●●	●●●	●	●●	●	
P13	●●	●●	●●●	●●●	●	●	●	
P14	●	●●	●●	●●●	●●	●●	●	●
D1	●		●●	●●	●●●	●●	●●	●●●
D2	●		●●	●●	●●●	●●●	●●●	●●●
H1	●●	●●	●●●	●●●	●	●	●	
H2	●●●	●●		●●	●			
H3	●●	●●	●	●	●●	●		
H4	●●●	●●	●	●●	●			
H5	●●●	●●●	●●●	●●●	●	●		
H6	●●●	●●●	●	●				
H7	●●●	●●●		●				
H8	●●●	●●●		●●				
W1	●●●	●●●	●●●					
W2	●●●	●●●	●●●					
W3	●●●	●●●	●●●					
W4	●●●	●●●	●●●					
W5	●●●	●●●	●●●					
W6	●●●	●●●	●●●					
Z1	●●●	●●●	●●					
Z2	●●●	●●●	●●●					
Z3	●●●	●●●	●●					
Z4	●●●	●●●	●●					
Z5	●●●	●●●	●●					

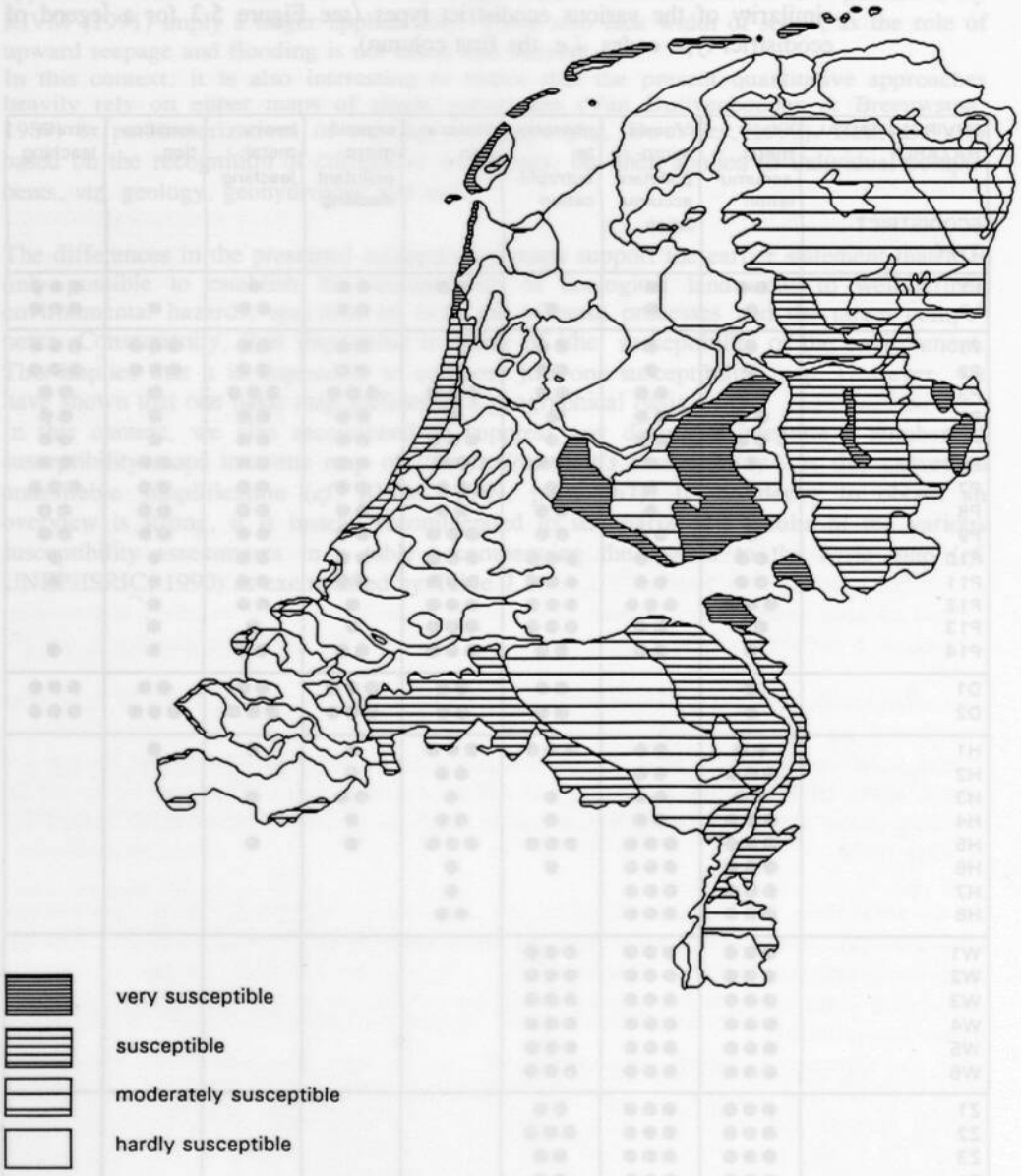

Figure 7.2 Susceptibility of ecodistricts to acidification of topsoil and shallow groundwater

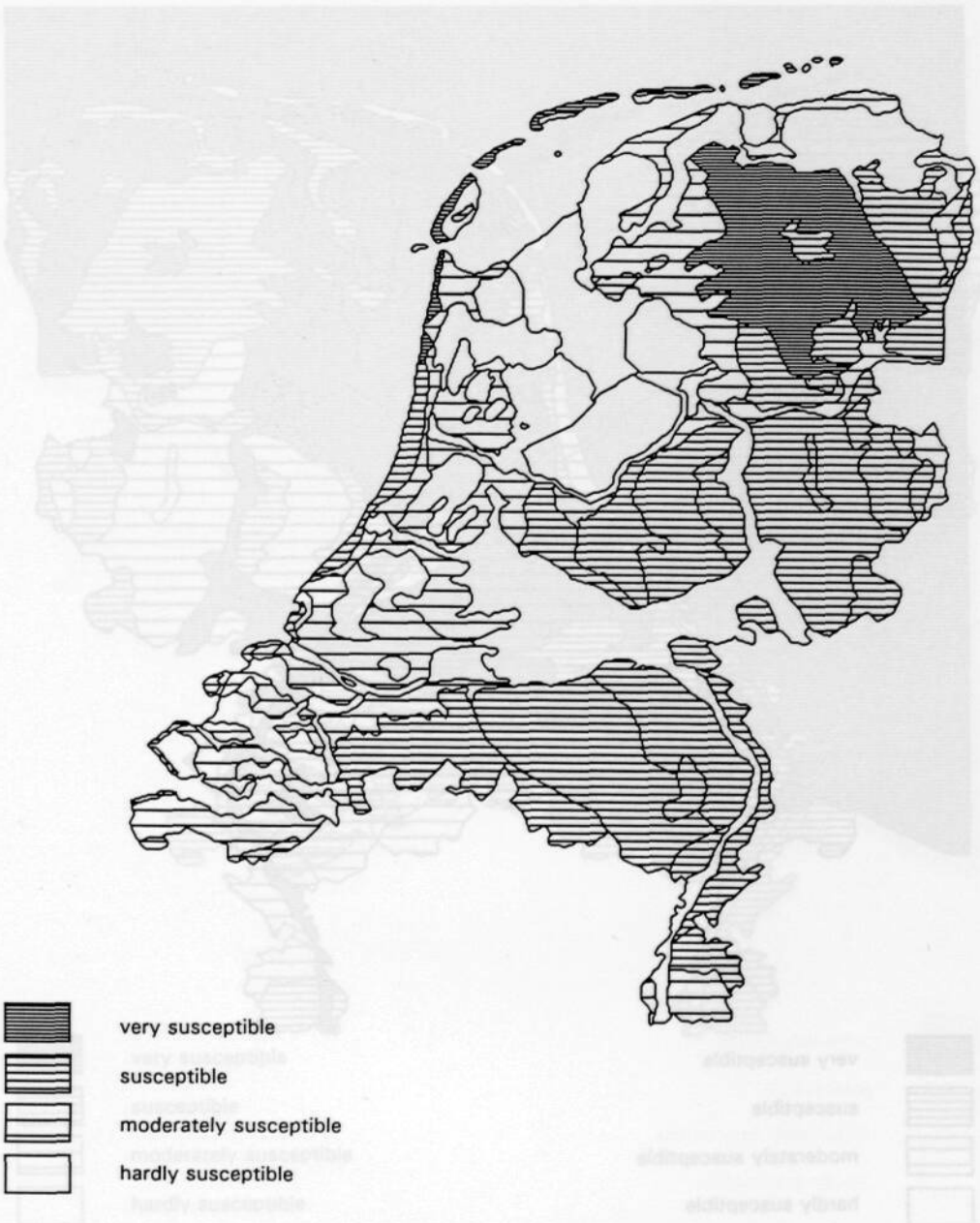

Figure 7.3 Susceptibility of ecodistricts to *leaching* of heavy metals to the groundwater

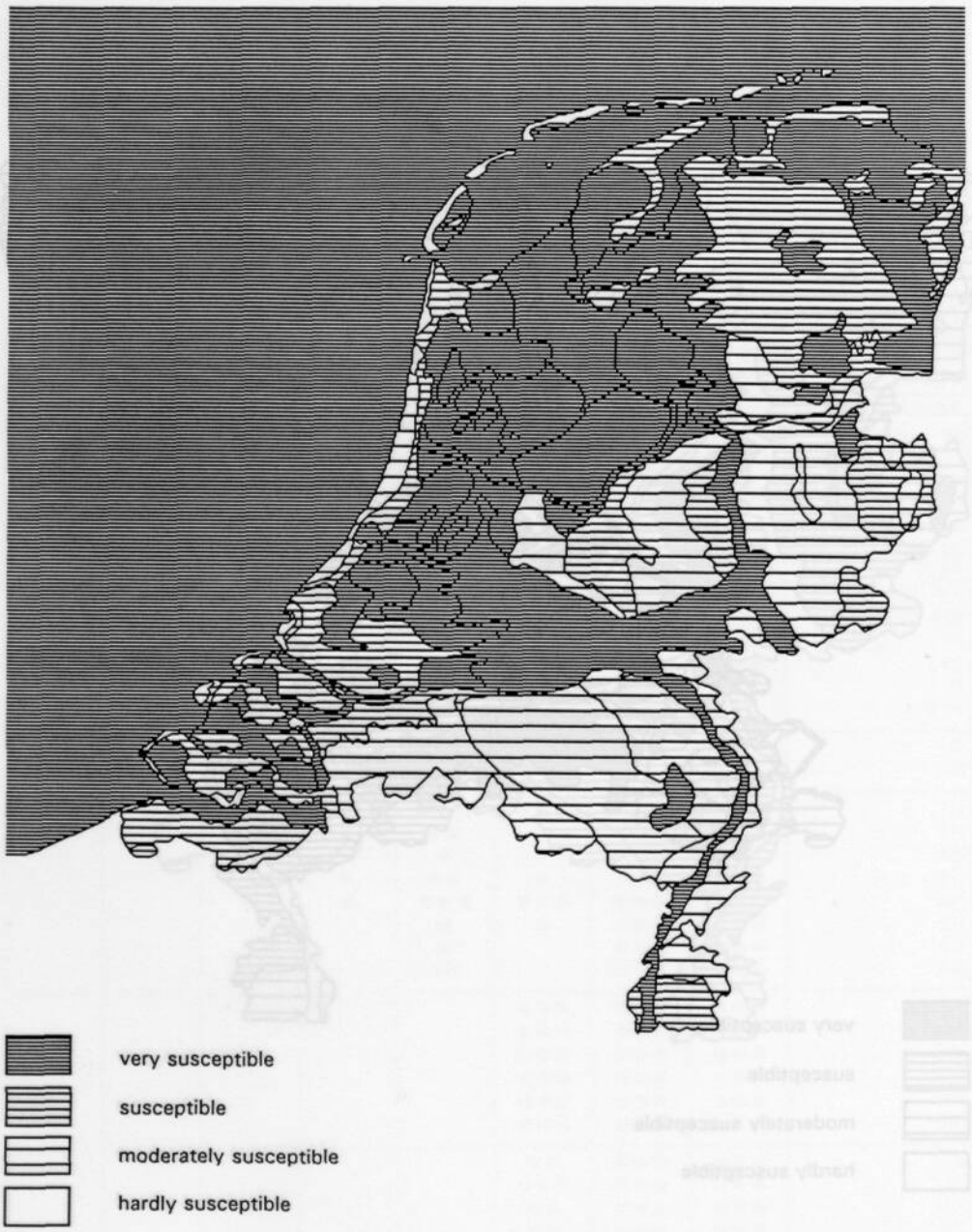

Figure 7.4 Susceptibility of ecodistricts to *accumulation* of heavy metals in the topsoil/sediment

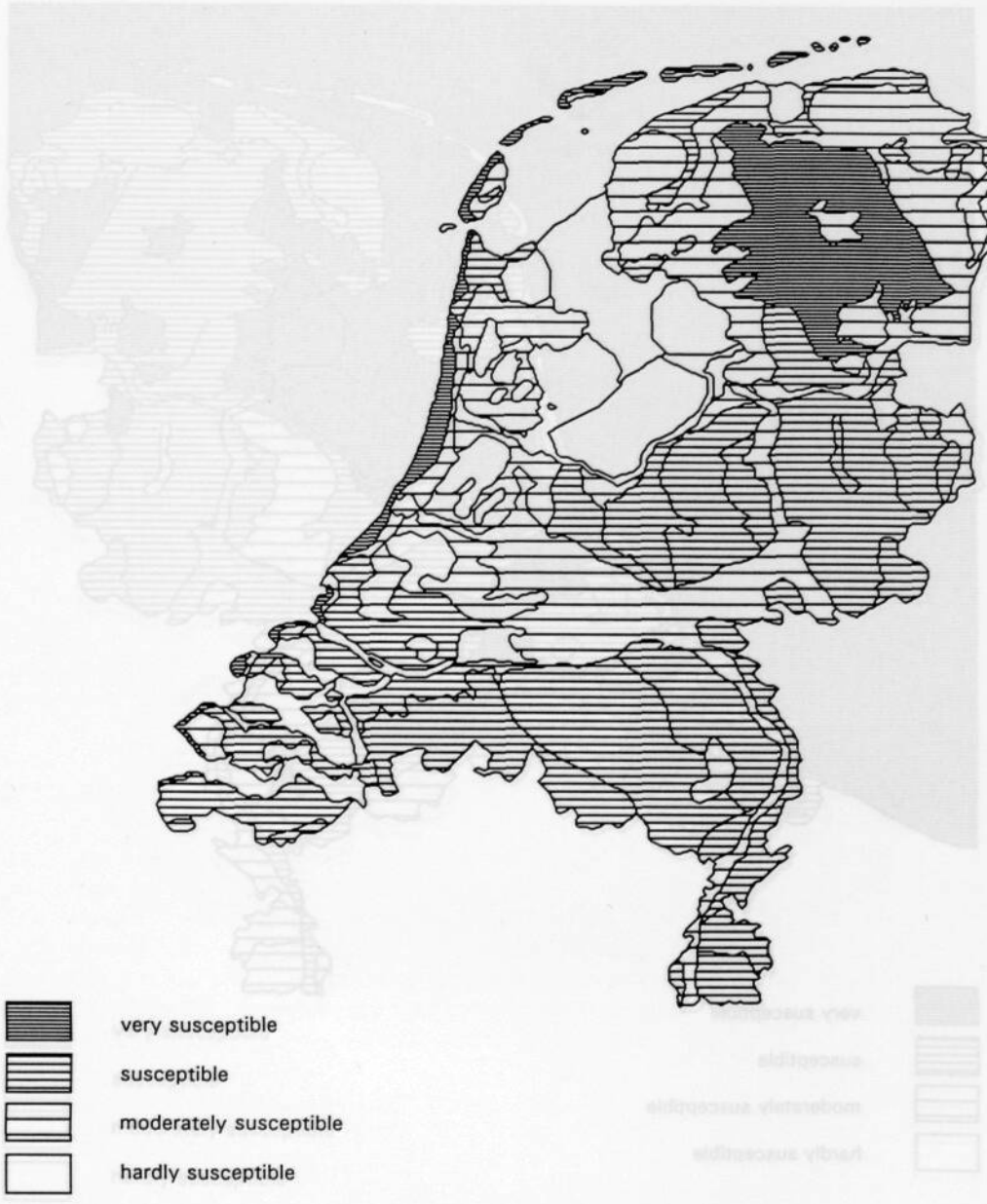

Figure 7.5 Susceptibility of ecodistricts to *leaching* of organic micropollutants to the groundwater

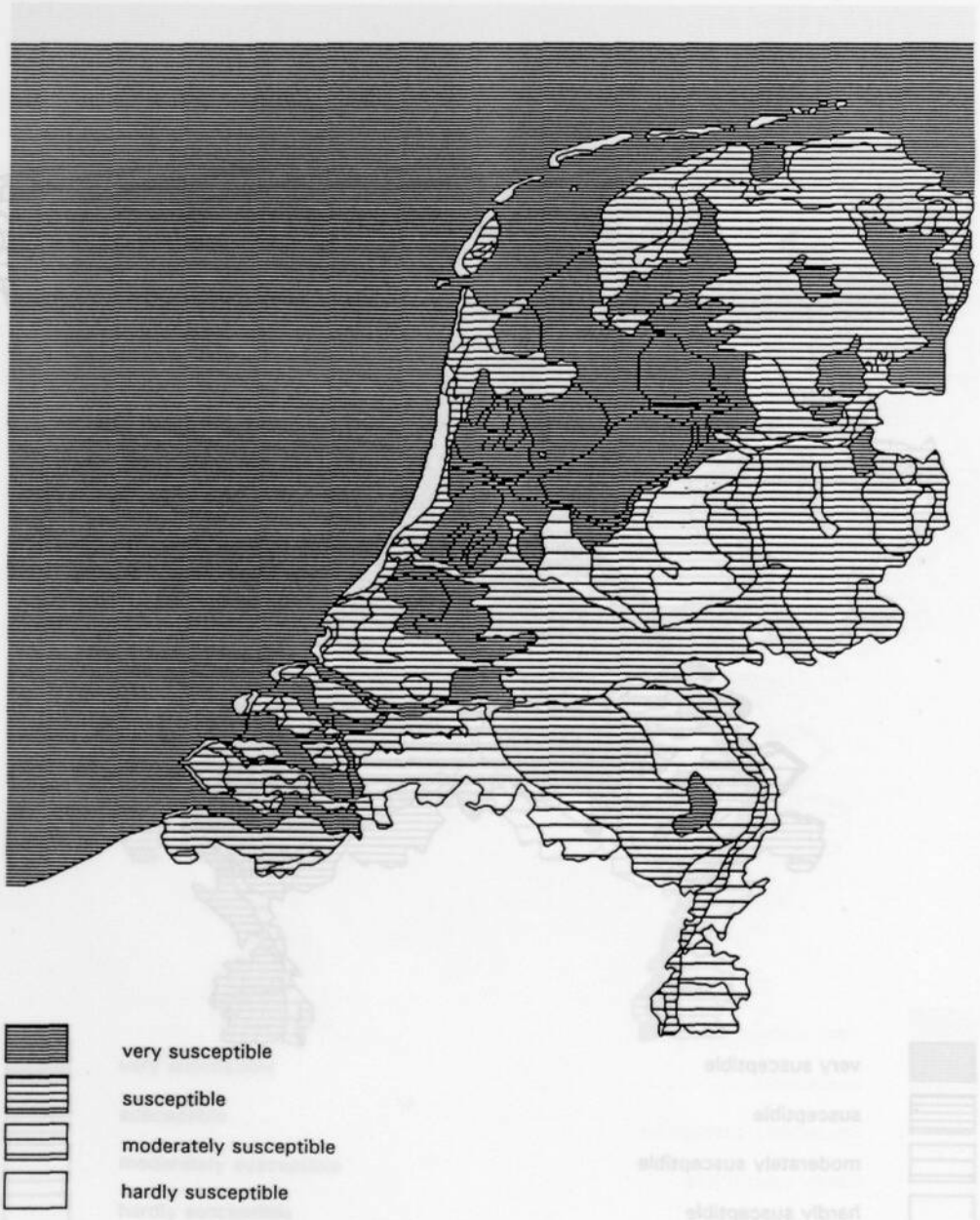

Figure 7.6 Susceptibility of ecodistricts to *accumulation* of organic micropollutants in the topsoil/ sediment

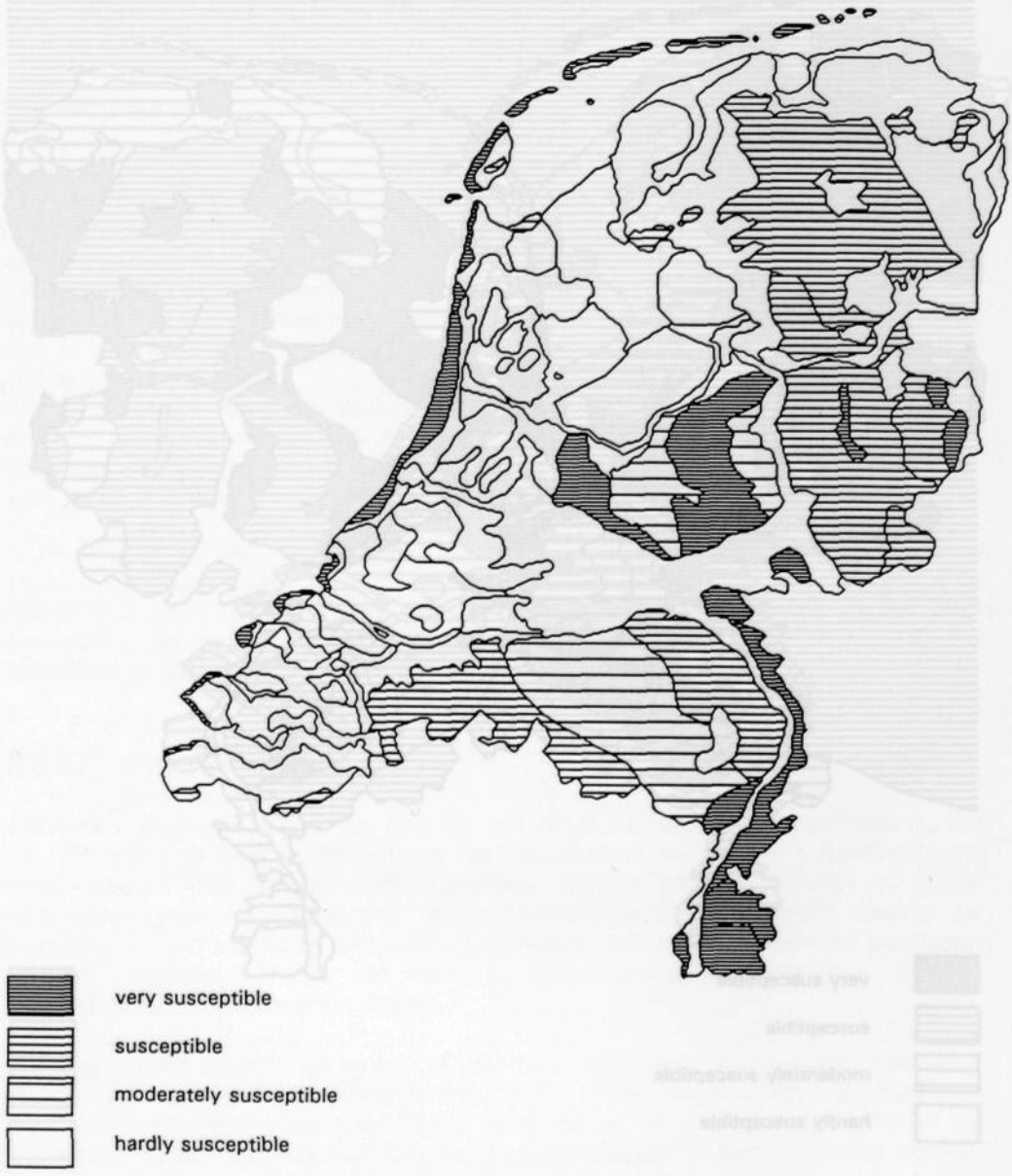

Figure 7.7 Susceptibility of ecodistricts to leaching of nitrate to the groundwater

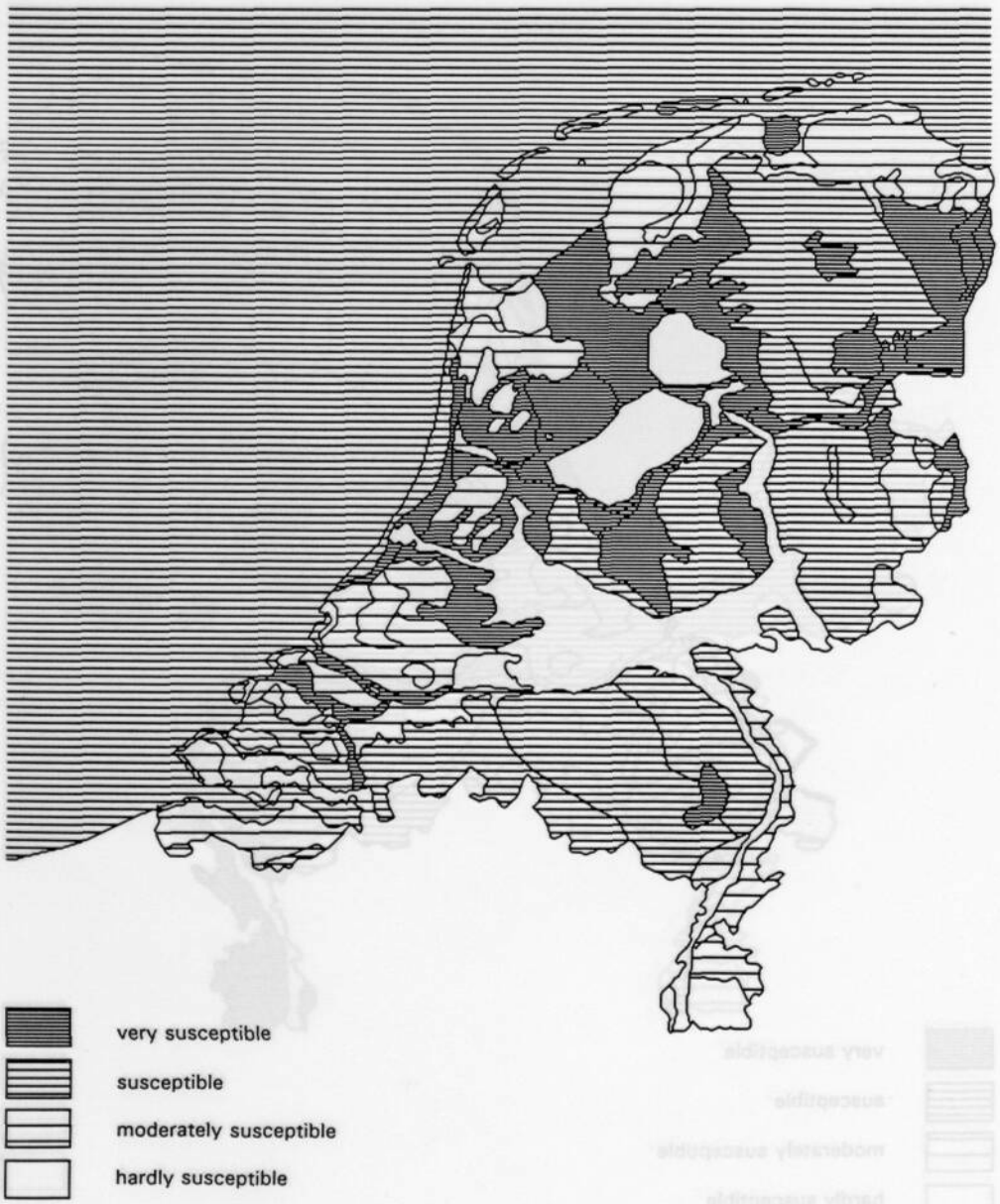

Figure 7.8 Susceptibility of ecodistricts to phosphate saturation of soils and/or phosphate eutrophication of surface waters

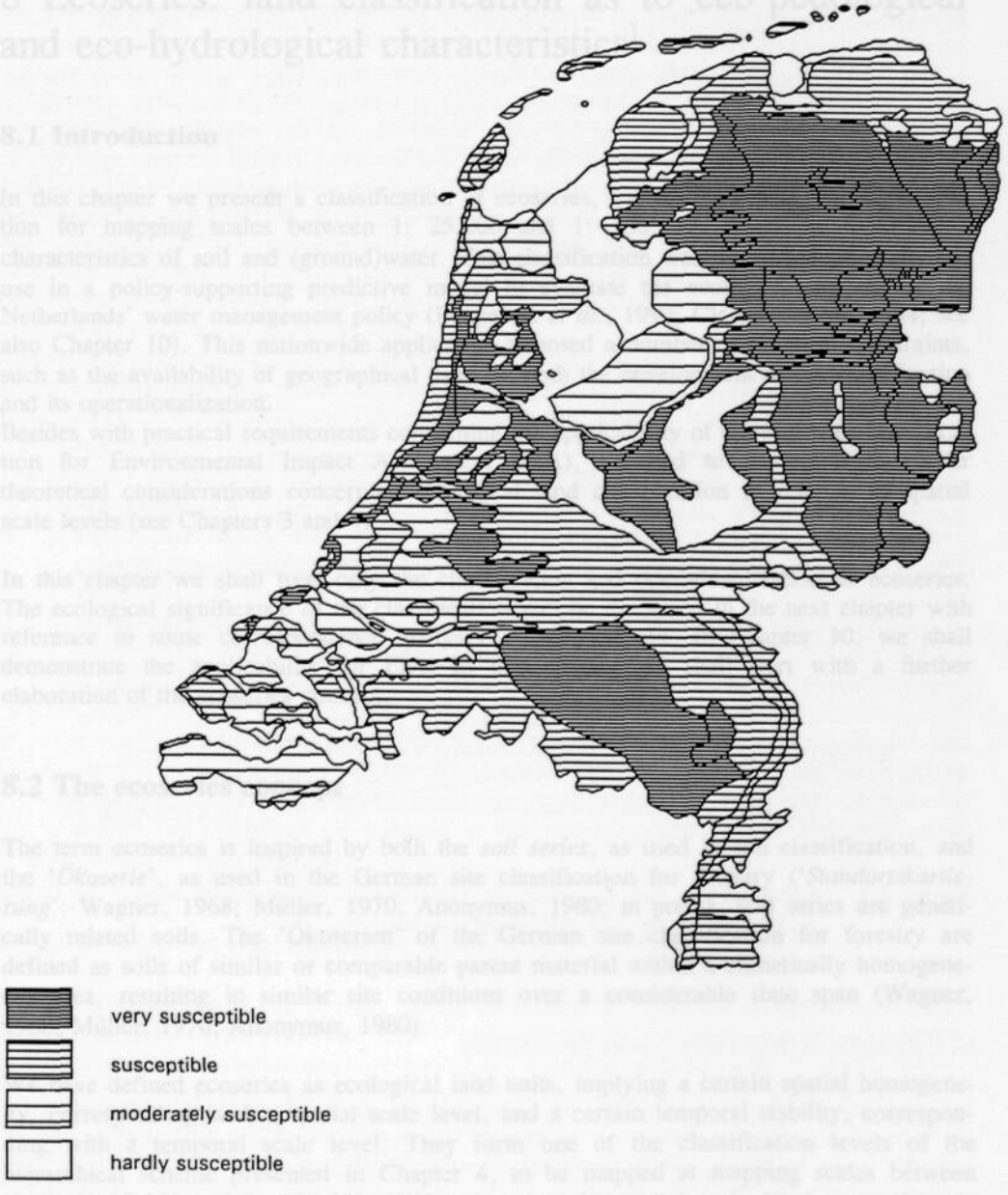

Figure 7.9 Susceptibility of ecodistricts to the effects of groundwater lowering

8 Ecoseries: land classification as to eco-pedological and eco-hydrological characteristics[1]

8.1 Introduction

In this chapter we present a classification of ecoseries, *i.e.* an ecological land classification for mapping scales between 1: 25.000 and 1: 100.000, based on conditioning characteristics of soil and (ground)water. This classification was developed primarily for use in a policy-supporting predictive model to evaluate the ecological impacts of the Netherlands' water management policy (Beugelink *et al.*, 1992; Claessen *et al.*, 1994; see also Chapter 10). This nationwide application imposed a number of practical constraints, such as the availability of geographical data, on both the development of the classification and its operationalization.
Besides with practical requirements concerning the applicability of the ecoseries classification for Environmental Impact Assessment (EIA), we had to comply with earlier theoretical considerations concerning ecological land classification in relation to spatial scale levels (see Chapters 3 and 4).

In this chapter we shall treat only the classification and operationalization of ecoseries. The ecological significance of the classification will be discussed in the next chapter with reference to some correspondence analyses we carried out. In Chapter 10, we shall demonstrate the applicability for EIA. In this chapter we shall start with a further elaboration of the ecoseries concept.

8.2 The ecoseries concept

The term ecoseries is inspired by both the *soil series*, as used in soil classification, and the '*Ökoserie*', as used in the German site classification for forestry ('*Standortskartierung*': Wagner, 1968; Müller, 1970; Anonymus, 1980; in press). Soil series are genetically related soils. The 'Ökoserien' of the German site classification for forestry are defined as soils of similar or comparable parent material within a climatically homogeneous area, resulting in similar site conditions over a considerable time span (Wagner, 1968; Müller, 1970; Anonymus, 1980).

We have defined ecoseries as ecological land units, implying a certain spatial homogeneity, corresponding with a spatial scale level, and a certain temporal stability, corresponding with a temporal scale level. They form one of the classification levels of the hierarchical scheme presented in Chapter 4, to be mapped at mapping scales between about 1: 25,000 and 1: 100,000. With respect to the spatial scale level, ecoseries are

[1]This chapter is an extended version of part of Klijn, F., C.L.G. Groen & J.P.M. Witte, 1996. Ecoseries for potential site mapping, an example from the Netherlands. *Landscape and Urban Planning* 35(1996): 53-70, including considerations expressed in Klijn, F. & R.W. de Waal, 1992. Ecologische bodemclassificatie; een pragmatische aanpak vanuit de standplaatsbenadering. *Landschap* 9/3: 175-187.

hence intermediate between ecotopes (mapping scales between appr. 1: 5,000 and 1: 25,000) and ecosections (mapping scales between appr. 1: 100,000 to 1: 500,000).

A classification of ecoseries should be based on characteristics which determine the pattern of ecosystems at the specified scale level and which remain stable for periods of tens of years when not disturbed by man. According to the guideline formulated in Chapter 4, soil, groundwater and surface water are supposed to yield the most appropriate classification characteristics at this scale level (Figure 8.1). Ecosections, which provide the larger spatial context for ecoseries, are determined primarily by geomorphology and (ground)water characteristics with soil being of minor importance (at 'group' level, *cf.* Section 4.3.3).

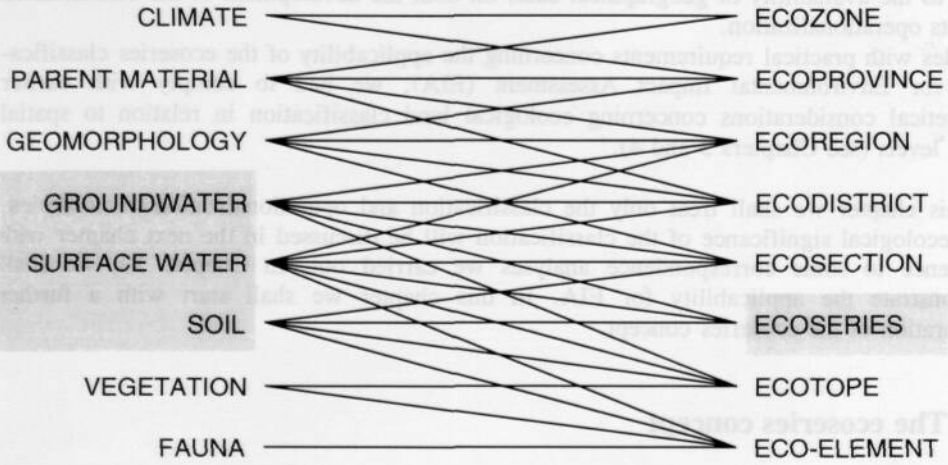

Figure 8.1 According to the guideline the classification of ecoseries should be based on (ground)water and soil characteristics.

8.3 Practical objectives and procedure

For any EIA the ecological effects of human actions must be quantified. Depending on the type of activities, a number of threatened characteristics of the environment can be selected, such as plants, birds, mammals, geomorphological values, cultural landscape characteristics, etc. In the present case, we shall focus on vegetation as the main threatened and valued ecosystem component, because it is the most closely related to the site factors affected by water management, eutrophication and acidification.

For predicting changes in the species composition of vegetations, a classification of *ecotopes* has been developed (Stevers *et al.*, 1987; Runhaar *et al.* 1987; Groen *et al.*,

1993; Runhaar & Udo de Haes, 1994). This classification is based on so-called *operative site factors* that determine the species composition of a vegetation (Figure 8.2). The ecotopes classification can be used to predict changes in the species composition due to changes in operative site factors (Runhaar & Udo de Haes, 1994).
In practice, many environmental hazards do not — or not directly — affect the operative site conditions, because abiotic processes counteract the effects (*e.g.* buffering or adsorption) or are triggered (*e.g.* accelerated mineralization or intensified natural acidification). It appears that especially for acidification, pollution with toxicants, eutrophication and lowering of the groundwater table these *conditioning* abiotic processes are a crucial interface for predictive modelling (see also Chapter 7). It is therefore necessary to understand the abiotic processes and to develop an ecological land classification which is also based on characteristics of soil, groundwater and other relevant abiotic components that control these processes.
Against this practical background the classification of *ecoseries* was developed, primarily as an interface between human interventions and/or immissions on the one hand, and biotic responses due to changing site conditions on the other.

The *interface* character of the ecoseries classification can, therefore, be specified in terms of a dual purpose:

- firstly, there should be a good *initial correspondence* between the ecoseries distinguished and the operative factors that determine the species composition of the vegetation, *i.e.* in *steady state*. This steady-state correspondence permits prediction of the recovery potential of a site, which depends largely on the 'natural', *i.e.* undisturbed, site conditions belonging to a certain combination of conditioning circumstances. It enables an estimate to be made of the 'potential vegetation' (see Chapter 9).
- secondly, ecoseries ought to be relevant for predicting *changes* in operative site factors as distinguished in the classification of ecotopes. This implies that the classification should cover characteristics that *control the processes*, in order to enable the prediction of changes in the operative site factors and subsequent changes in the vegetation.

8.3.1 Operative and conditioning site factors

By taking vegetation as the primary living component of terrestrial ecosystems, we may start investigations into which site factors[2] are ecologically the most important. Probably, the most basic site factors are rooting space, moisture and nutrients (Russell, 1973). These factors, however, can be subdivided and/or related to other site factors of more indirect importance, such as pH (affecting both nutrient availability and moisture exchange between plant and soil), salinity (affecting moisture exchange), temperature, aeration, etc. Following the causal relationships even further back we encounter soil texture, soil structure, organic matter content, groundwater levels, etc., or even

[2]Site factors comprise both *conditions sensu* Begon *et al.* (1986, page 41), which cannot be consumed by organisms and are never limitative, *e.g.* temperature, and *resources sensu* Begon *et al.* (1986, page 75), which may be limitative, *e.g.* nutrients.

geomorphological position, composition of parent material, altitude, etc.
In fact, depending on our efforts we may find that a vegetation's site is 'determined' by anything between merely rooting space, water and nutrients on the one hand, and a wealth of climatic factors (radiation, light, temperature, frost, precipitation, etc), parent material (degree of consolidation, chemical composition, content of weatherable minerals, etc.), physiographic factors (exposition, altitude, erosion, etc.) groundwater, surface water, and soil factors on the other hand. Ideally, sites — or ecotopes for that matter — should of course be defined very narrowly by using as many relevant factors as classification characteristics as possible, but for practical reasons we must limit ourselves to a reasonable number of easily detectible characteristics to achieve a comprehensive classification (Kilian, 1981, page 137).

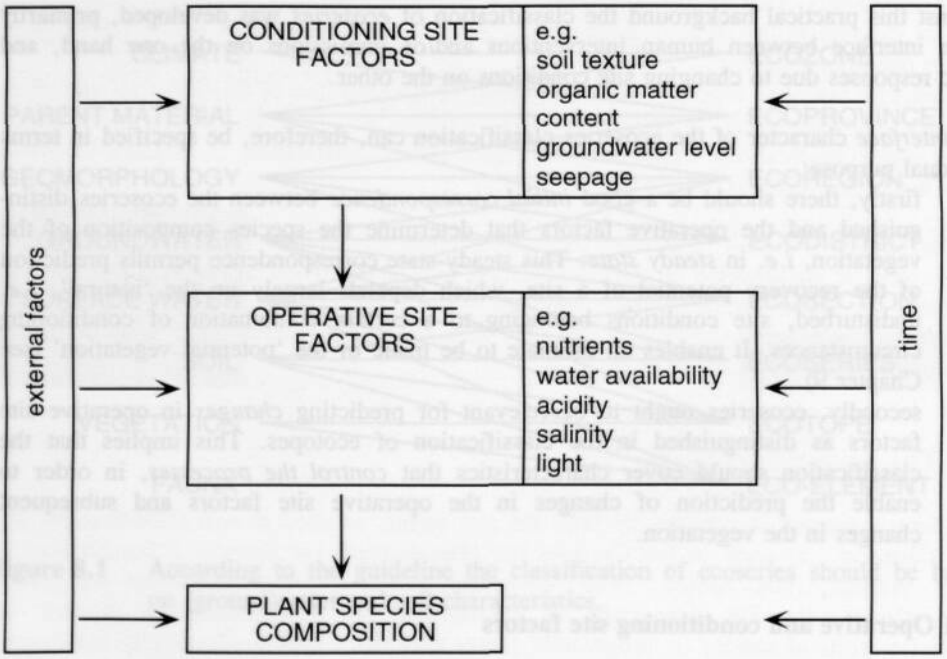

Figure 8.2 The difference and relationship between relatively conditioning and relatively operative site variables, successively affecting the plant species composition of vegetations.

Instead of developing one classification on the basis of all ecologically relevant characteristics, we argued in Chapter 4 that we should rather develop a number of simpler, related classifications for different spatial scale levels. In search of the most appropriate classification characteristics for the scale levels of ecotopes and ecoseries it may help to distinguish between physiological, operative, conditioning and topological site factors (combined after distinctions by, among others, Jenny, 1941; Van Wirdum, 1979; 1986; and Kemmers, 1993). In fact, the distinction is not so clear but rather relative. Therefore,

we shall only retain the difference between *relatively conditioning* and *relatively operative* factors (figure 8.2) as advocated by Van Leeuwen and Van Wirdum (*cf.* Van Wirdum, 1979; 1986; also Zonneveld, 1995, page 61), two categories which closely correspond, respectively, with Hills' (1953) 'potential features' and 'available features'.
This distinction relates to the hierarchy of processes according to causality, as described in Section 4.2.1. Conditioning site factors have indirect effects on vegetation only, whereas operative site factors have more direct effects on a vegetation's species composition. Thus, the conditioning site factors partly control the operative factors. Because the hierarchy of processes is also related to a hierarchy of patterns, the larger spatial units defined by conditioning factors provide a relatively homogeneous spatial context for the smaller units defined by operative site factors. If, for example, a site classification's validity is confined to the Netherlands' country borders we, despite a gradual transition from more Atlantic in the southwest to more continental in the east, may neglect climatic differences. Similarly, on the other side of the spectrum we can neglect minor variations in base saturation or soil temperature. Thus, the site concept is confined to ecosystems of a certain spatial extent ranging from about 2,500 m^2 to sveral square kilometers in very homogeneous landscape, approximately corresponding with the size of plant communities as distinguished in vegetation science.

8.3.2 Ecotopes as defined by operative site factors: a first point of departure

Stevers *et al.* (1987) developed an *ecotope* classification for the Netherlands based on a selection of the most determinant operative site factors (see also Groen *et al.*, 1993; Runhaar & Udo de Haes, 1994). They defined ecotopes as '*spatial units which are homogeneous as to vegetation structure, succession stage, and operative site factors which determine the species composition of the vegetation*', and selected the following *abiotic* site factors as the main classification characteristics, each divided into three to four classes:

- salinity
- moisture regime
- nutrient availability
- acidity

When combined with *vegetation structure* classes, the classes for these abiotic classification characteristics define the main ecotope types. Additionally, a number of subtypes were distinguished with excessive dynamics of flooding or sand drift or on odd substrates such as brick walls. These subtypes will not be taken into account here.
By clustering all the main ecotope types that differ only in vegetation structure but with equal classes for *abiotic* site factors, we obtain an abiotic *site diagram* as presented in Figure 8.3. The site classification thus achieved is very similar to site classifications which are frequently used for forestry in various parts of the world (*e.g.* Arnborg, 1964; Bannink *et al.*, 1973; Anonymus, 1980; Kilian, 1981; Karrer & Kilian, 1990). The abiotic site types may also be regarded as equivalent to the concept of 'physiotope' (*cf.* Leser, 1991, pages 148 and 256), a concept which is, however, also used for larger ecological land units classified by conditioning characteristics (*cf.* Leser, 1991, page 338),

e.g. by Farjon (1987; *cf.* also Harms *et al.*, 1991, page 36-42, and Harms & Roos-Klein Lankhorst, 1994, page 91-103), *i.e.* more or less identical to our concept of ecoseries (see Section 8.3.5).

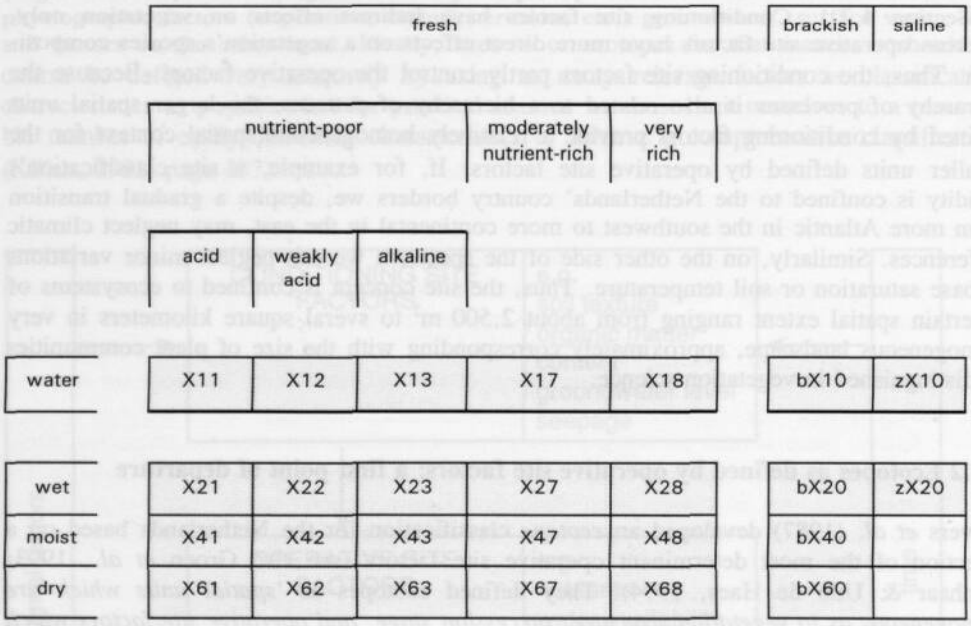

Figure 8.3 Operative abiotic site types, as derived from an aggragation of ecotope types with corresponding abiotic site characteristics ('poor' and 'rich' concern nutrient availability). 'X' stands for 'abiotic site type'. The vertical axis corresponds with moisture classes (first figure of code), the horizontal axis with the combination of nutrient availability and acidity (second figure of code), as well as salinity classes (indicated with a prefix when not fresh).

Knowledge of the operative site factors allows estimation of the 'potential vegetation' in terms of series of successional stages related to species and/or plant communities *within a confined geographical region* (*cf.* the German '*Wuchsgebiet*' and '*Wuchsbezirk*'; Anonymus, 1980; in press). Thus, for example, pioneer vegetation, grassland, tall herbaceous vegetation, shrubland and woodland can be distinguished on sites with similar abiotic site conditions.

In practice, site classifications on the basis of merely operative site factors also bring along a number of problems. Firstly, there are no nationwide maps available on operative factors, thus prohibiting nationwide analyses. Secondly, operative site factors may change very rapidly as a result of human influence, implying that the present state does not

necessarily correspond with the pristine or potential state. Thirdly, most environmental hazards do not directly affect the operative site factors, because abiotic processes counteract the effects (buffering, leaching, adsorption) or are triggered (accelerated mineralization or intensified natural acidification). In fact, the results of acidification, eutrophication or lowering of the groundwater table cannot be understood without taking into account these processes which are controlled by conditioning site factors.
We may therefore conclude that for a nationwide overview we should concentrate on relatively stable site factors, and also that for a full understanding of a site's responsiveness to environmental change, we must take into account conditioning site factors as well. Thus, we are back to the motive of developing an *ecoseries* classification in connection with the existing ecotopes classification, but based entirely on relatively stable, *conditioning* site factors.

8.3.3 From operative site factors to conditioning factors

The site factors used for classifying ecotopes are ecologically the most decisive, but they are relatively easily changed and subject to rapid natural fluctuations and human interference. Also, they are difficult to establish for large areas or nationwide.
However, the operative site factors are dependent on the interaction of a number of more conditioning site factors such as soil texture, organic matter content, $CaCO_3$-content, groundwater level and the chemical composition of upward seepage. In fact, we are again dealing with correlative complexes (see Chapter 7), where the correlations are not one to one, but still significant.
Against this background we can define ecoseries as:

> *spatial units which are homogeneous as to abiotic ecosystem characteristics that condition the operative site factors relevant for the species composition of the vegetation.*

To define potentially relevant classification characteristics for ecoseries, we arranged the knowledge about the relationships between operative site factors and conditioning site factors found in comprehensive literature (*e.g.* Russell, 1973; Scheffer & Schachtschabel, 1976; De Bakker & Locher, 1990) (Table 8.1). The conditioning factors are generally characteristics of ecosystem components which are hierarchically placed above the vegetation, *i.e.* soil and (ground)water.
With the characteristics that control the relevant operative site factors in steady-state conditions we cover the first requirement of the ecoseries classification. However, a dual purpose was formulated; the ecoseries should also enable predictions to be made of the impacts of human activities, *i.e.* changes in operative site factors. We shall consider this point in the next section.

Table 8.1 The relationship between operative site factors as recognized in the ecotopes classification focusing on vegetation and the conditioning ecosystem characteristics that primarily control these operative site factors.

OPERATIVE SITE FACTORS / CONDITIONING CHARACTERISTICS	salinity	moisture availability	nutrient availability	acidity
parent material/ peat type		●	●	●
texture		●	●	
age of soil			●	●
pedogenesis			●	●
profile differentiation/ soil depth		●		
organic matter content		●	●	
$CaCO_3$-content				●
groundwater level (fluctuations)		●		●
location (inundation/ sea-spray)	●		●	●
cation content of upward seepage			●	●
salinity of upward seepage	●			

8.3.4 Ecosystem characteristics that control the response to environmental hazards

The response of ecosystems to a number of environmental hazards may depend on abiotic processes. This is true for the effects of deposition of acid-producing substances, toxic substances or eutrophying substances, as well as for changes in local hydrology. It relates precisely to the environmental hazards and connected processes discussed in Chapter 7 (compare also Figure 7.1), which implies that we are interested in abiotic processes such as acid neutralization, mineralization, denitrification or phosphate fixation. We found that these processes are controlled by abiotic ecosystem characteristics, such as clay content, organic matter content or upward seepage intensity and quality. Partly, they involve characteristics other than the determinant characteristics for steady-state situations.
As we want the ecoseries classification to be relevant for ecological impact assessment, we must also take these process-controlling factors into account. Also, against this background, the definition of ecoseries may be extended to read:

> *spatial units which are homogeneous as to those abiotic ecosystem characteristics that condition the operative site factors determining the species composition of the vegetation, and/or with regard to those abiotic ecosystem characteristics that control the effects of environmental hazards.*

Table 8.2 Conditioning ecosystem characteristics considered to be the most relevant controlling factors for the abiotic processes related to some environmental themes, with a focus on soil and groundwater characteristics relevant for terrestrial situations.

ENVIRONMENTAL THEMES / CONDITIONING CHARACTERISTICS	acidification	pollution	eutrophication	desiccation
texture	●	●	●	●
content of weatherable silicates	●			
primary $CaCO_3$-content	●	●	●	●
depth of decalcification	●	●	●	
organic matter content	●	●	●	●
type of organic matter			●	●
Fe- and Al(hydr)oxides	●		●	
groundwater level (fluctuations)	●	●	●	●
seepage (direction and rate)	●	●	●	●
cation content of upward seepage	●	●	●	●
salinity of upward seepage	●	●	●	●

In Chapter 7 we have already defined the processes relating to acidification, pollution with toxicants, eutrophication and desiccation. We also traced the relevant parameters, *i.e.* the controlling factors, for these processes. This resulted in the list of Table 7.1, which we shortened to constitute Table 8.2, which covers only the factors of importance for terrestrial situations[3].

8.3.5 Classification characteristics and classes

From the foregoing considerations we derived a large number of relevant factors. In theory, all can be considered as classification characteristics. For a comprehensive ecoseries classification this would be very impracticable indeed. Therefore, a reduction in number was considered necessary to achieve a practicable classification.
In Table 8.3 we arranged our overall list of relevant conditioning ecosystem characteristics resulting from the combination of Tables 8.1 and 8.2 into six groups on the basis of correlative and causal relationships, also relating them to the main ecosystem components soil and (ground)water. Subsequently, we selected classification characteristics from each of these groups that were supposed to cover the others within a group.

[3]We limit ourselves to an ecoseries classification for terrestrial situations, comprising only very small surface waters that can be regarded as inclusions in terrestrial map units.

Table 8.3 Groups of conditioning ecosystem characteristics that are closely related, *i.e.* almost synonymous or correlated through causal relationships; the further to the right, the more (relatively) dependent. The range used for the selection of classification characteristics is indicated by ▾.

SOIL	▾ ▾
group 1	parent material texture peat type age of parent material content of weatherable silicates pedogenesis Fe- and Al-(hydr)oxydes type of organic matter
group 2	profile differentiation (of parent material/texture)
group 3	organic matter content
group 4	$CaCO_3$-content
(GROUND)WATER	▾ ▾
group 5	groundwater level (fluctuations)
group 6	location (inundation/ sea-spray) seepage (direction and rate) cation content of upward seepage salinity of upward seepage

The classification characteristics were selected on the basis of the following theoretical and practical considerations, partly treated in the previous sections:

- firstly, there should be no doubt about the ecological relevance, which is to be considered as an unambiguous relationship with (either) the operative site factors of the ecotope classification and/or a crucial role in controlling the abiotic processes determining their response;
- secondly, the classification characteristics should be as far as possible independent of one another, but meanwhile cover the whole set;
- thirdly, the characteristics should be related unambiguously to available geographical data. This consideration is primarily a practical one, because the classification should be applicable for EIA with respect to regional and national environmental management policy.

More specifically, the selection per group was based on the following considerations:

- *Group 1*: Parent material and texture are complementary characteristics, in the sense that texture does not cover peat soils, whereas for mineral soils we are primarily interested in the texture. The peat type is a specification of the parent material after the dominant component species (reed-sedges peat and alder/birch peat: in relatively eutrophic sites; versus sphagnum peat: oligotrophic sites). For mineral soils the age is important, because it determines the degree of weathering and soil genesis. The content of weatherable silicates, in contrast, can be regarded as a function of the parent material depending on the age; consequently, it can be derived from the others, whereas it is, in contrast, a controlling factor for soil genesis itself. Soil genesis is further dependent on the texture and the groundwater regime (see groups 5 and 6).
  Both the iron and aluminium (hydr)oxide content and the type of organic matter are a function of the pedogenesis. Both can be estimated fairly accurately if the pedogenesis is known in relation to the parent material and groundwater regime (see groups 5 and 6).
  As the classification characteristic we select the couple *parent material/texture*, as this enables a fair derivation of the other relevant factors. In the (qualitative) class distinction both the peat type and the age ought to be covered.

- *Group 2*: The profile differentiation is a further specification of the previous characteristic. In fact, it concerns the vertical layering of various types of parent material (peat type and/or texture).
  We shall regard *profile differentiation* as a separate classification characteristic.

- *Group 3*: The organic matter content results from soil genetic processes, whereas peat formation is regarded as a geogenetic process. Therefore, peat types are covered by the characteristics parent material and profile differentiation, whereas *organic matter content* is selected as a separate classification characteristic for mineral soils.

- *Group 4*: The $CaCO_3$-content is a function of the parent material and soil genetic processes. The age of a soil is very important in this context, as well as the groundwater regime. The $CaCO_3$-content is one of the most important determinants for the acidity of a site, which is regarded as an operative site factor in the ecotopes classification, but which also has a large influence on relevant abiotic processes. We regard the *$CaCO_3$-content* as sufficiently important to select it as a separate classification characteristic.

- *Group 5*: Groundwater level fluctuations are directly relevant for the moisture availability, in interaction with the soil texture, and for the acidity of a site. Indirectly, they are important because they are, *inter alia*, a controlling factor for the redox conditions that are relevant for a number of abiotic processes, for example, denitrification, cation exchange or phosphate fixation. Because redox conditions are difficult to establish, we derive them from the conditioning factors that are covered by groups 1, 2, 3 in combination with the *groundwater table*

class, which we selected as a classification characteristic. As we use alternative classes from those of De Vries & Wallenburg (1990), we use this more general term instead of groundwater level class.

- *Group 6*: The geographical location refers to the geomorphological setting, which determines the flood-proneness and the vertical groundwater movements (infiltration/exfiltration). It may be a relevant classification characteristic for ecosections, *i.e.* at the next higher classification level, but it is in fact not specific enough to really consider it as a classification characteristic for ecoseries. We do, however, use it to distinguish periodically flooded ecoseries, but without further differentiating these.

  Upward seepage can be considered as a class within the more encompassing concept of seepage, which may be split up according to direction and rate. As downward seepage or very low rates of upward seepage do not directly nor significantly influence operative site conditions, we consider upward seepage a sufficiently adequate classification characteristic.

  Moreover, only upward seepage significantly influences the rooting zone by its chemical composition. We could, for example, specify the upward seepage quality in relevant quality classes by its salinity and cation content (Van Wirdum, 1980; Klijn, 1989). Accordingly, we selected *salinity/cation composition of upward seepage* (if present) as a classification characteristic.

For the time being, we shall also use *enrichment with iron oxides* as an additional classification characteristic, because, firstly, nationwide data on upward seepage (group 6) are often difficult to obtain, whereas enrichment with iron oxides often occurs in places with present or past upward seepage.

Thus we determined the following seven classification characteristics:

1 parent material/texture
2 profile differentiation (with respect to 1)
3 organic matter content
4 $CaCO_3$ content
5 enrichment with iron-oxides
6 groundwater level
7 salinity/cation composition of upward seepage

These are mainly characteristics of soil (1 to 5) and groundwater (6 and 7). This is fully concordant with the ecoseries concept as used in German literature. For example, Müller (1970) speaks of ecoseries as generalized soil types in combination with drainage conditions which from a forestry point of view constitute similar sites. However, both Wagner (1968) and Müller (1970) use the drainage condition as adjective, thus forming *subtypes*, whereas we selected the groundwater characteristics to classify *types*.

For each classification characteristic a limited number of classes were defined, again in relation to the ecotopes classification and the available geographical data (see the next section). The combination of the classes yielded the *ecoseries classification*, which can be

regarded as a combination of an ecological soil classification, an ecologically relevant classification of groundwater levels and an eco-hydrological classification of seepage.
The provisional classification thus formed was checked for ecological relevance and likeliness to be encountered in reality. Only the meaningful and realistic combinations have been retained.

8.4 Practical operationalization

With the above, deductive approach we obtained a provisional ecoseries classification with respecrt to ecologically relevant soil and groundwater characteristics. The next step was an examination of the possibilities to obtain practically applicable maps for the desired applications on the basis of existing geographical data, preferably by merely converting, generalizing and combining the legends of existing maps of soil and groundwater in accordance with the provisional classification.

More specifically, the nationwide analyses for the Netherlands' water management (Beugelink *et al.*, 1992; Claessen *et al.*, 1994), for which the ecoseries classification was primarily developed, required operationalizing the ecoseries classification in the form of a nationwide database on a 1 km^2 gridcell basis. This practical requirement followed from the decision to acquire all relevant geographical data and to carry out all further calculations for 1 km^2 gridcells.
The intended applications therefore required the conversion of existing maps and/or geographical databases on soil and groundwater characteristics according to the provisional classification.

8.4.1 Existing geographical data on soil and groundwater

As the majority of soil classifications and mappings, the Netherlands' soil classification and mapping developed along two mainstreams at the same time. The first mainstream is guided predominantly by applications, in the context of, for example, land evaluations for agriculture or forestry. The second mainstream results from more fundamental scientific interest. Consequently, many soil classifications (*e.g.* US Soil Taxonomy of the Soil Survey Staaf, 1975; legend to the Soil Map of the World by FAO, 1988), including the Netherlands' soil classification (De Bakker & Schelling, 1966; 1989) and maps (Steur *et al.*, 1985; Steur & Heijink, 1987), are mixtures of classifications according to soil characteristics that are relevant for land utilization ('functionally guided classification' *sensu* Zonneveld, 1994, page 28) on the one hand, and pedogenetically inspired classifications ('genetic guiding principle' *sensu* Zonneveld, 1994, page 28) on the other. The 1: 50,000 soil map, for example, contains legend units which result from combining pedogenetically important phenomena, such as podzolisation or illuviation, with soil characteristics that are more important for agriculture or forestry, such as texture, stoniness or drainage class.

For our purpose, the 1: 250,000 soil map (Steur *et al.*, 1985) might have qualified because of its nationwide coverage and the availability in digitized form. However, it was

considered insufficiently accurate in the spatial dimension, and the legend insufficiently ecologically relevant. Small but ecologically extremely relevant variations in $CaCO_3$-content, clay content in sandy soils or groundwater levels were discarded, whereas pedogenetical characteristics were retained in this 'generalization' of the 1: 50,000 soil map.

In contrast, the 1: 50,000 soil map (Steur & Heijink, 1987; published in about 60 sheets/volumes) was found much more appropriate. It contains a wealth of information, including information on parent material, geomorphology and groundwater level fluctuations. As to the latter, it contains, in comparison to international standards, very specific information on groundwater level fluctuations (Steur & Heijink, 1987; De Vries & Van Wallenburg, 1990). In fact, it can be regarded as a partly 'integrated' land classification rather than a soil classification, primarily because it is physiographically based.

Consequently, but unfortunately, the 1: 50,000 soil map contains more than 2,000 legend units which makes it hard to handle for nationwide applications. However, it is obvious that the large number of legend units can be reduced substantially by discarding all information irrelevant for our purposes.
In this context, a first 'ecological' generalization had already been carried out for the Landscape Ecological Mapping of the Netherlands (LKN; Canters *et al.*, 1991; Bolsius *et al.*, 1994). For this geographical database, De Waal (1992) reduced the number of legend units for soil to 230, meanwhile adding ecologically relevant data, for example on recurrent flooding or salt-spray. This LKN soil database gives the surface area (as a percentage) of all combinations of soil and groundwater level classes occurring in a 1 km^2 gridcell. As it comprises information on 6 of the 7 classification characteristics for ecoseries, it was used as the primary source to establish a 1 km^2 gridcell database on ecoseries for the Netherlands by further generalization.

With respect to the seventh classification characteristic, the quality of upward seepage, data are available in LKN as well (Klijn, 1989). However, these have only recently been connected with the soil data, as this posed substantial practical problems requiring a large number of decision rules and substantial data handling (Klijn *et al.*, 1996). In the meantime, we presumed the combination of certain soil characteristics, such as the enrichment with iron oxides, and specific groundwater level classes to be sufficiently indicative for upward seepage. This implies that the classification we are presently discussing is only the second approximation, which is the reason for addressing it as the 2.0 version.

8.4.2 Ecoseries 2.0

The second approximation of the ecoseries classification consists of 49 soil units (Table 8.4) with 6 groundwater table classes (Table 8.5). In practice, only 186 relevant combinations of soil units with groundwater table classes were found to exist in the LKN database (Table 8.6). Additionally, legend units such as surface water (A01 and GWT class 0) and urban area (A02) have been recognized.

Table 8.4 The 49 ECOSERIES-SOIL units distinguished on the basis of the selected classification characteristics and actual occurrence in the Netherlands.

CODE	DESCRIPTION
V01	primarily oligotrophic peat, not mineralized (no Fe)
V02	primarily oligotrophic peat (partly mineralized, no Fe)
V03	primarily oligotrophic peat enriched with iron
V04	primarily meso-eutrophic peat (no Fe, no sand cover)
V05	primarily meso-eutrophic peat with sand cover (no Fe)
V06	primarily meso-eutrophic peat enriched with iron (no sand cover)
V07	primarily meso-eutrophic peat enriched with iron and with sand cover
V08	peat covered with clay (no Fe)
V09	peat covered with clay and enriched with iron
V10	peat on sand (no sand cover)
V11	peat on sand with sand cover
K01	non-calcareous light clays and loam (not on sand or peat)
K02	non-calcareous light clays and loam on sand
K03	non-calcareous heavy clays (not on sand or peat)
K04	calcareous light clays and loam (not on sand or peat)
K05	calcareous light clays and loam on sand
K06	calcareous heavy clays (not on sand or peat)
K07	slightly calcareous light clays and loam on peat
K08	slightly calcareous heavy clays on peat
K09	acid sulphur clays
K10	decalcified chalk weathering (clay or loam)
K11	calcareous chalk weathering (clay or loam)
K12	old strongly weathered clays (no sand cover)
K13	old strongly weathered clays with sand cover
K30	periodically flooded clay soils (not differentiated)
Z01	peaty sands (no Fe, no clay cover)
Z02	peaty sands, enriched with iron (no clay cover)
Z03	peaty sands with clay cover (no Fe)
Z04	non-calcareous sands with organic topsoil (no Fe)
Z05	non-calcareous sands with organic topsoil enriched with iron
Z06	calcareous sands with organic topsoil (no Fe)
Z07	non-calcareous sands with thin topsoil (not loamy, no Fe)
Z08	calcareous sands with thin topsoil, enriched with iron (not loamy)
Z09	loamy non-calcareous sands with thin topsoil (no Fe)
Z10	calcareous sands with thin topsoil (not loamy, no Fe)
Z11	loamy calcareous sands with thin topsoil (no Fe)
Z12	non-calcareous sands, rich in organic matter (not loamy)
Z13	loamy non-calcareous sands, rich in organic matter
Z14	non-calcareous sands without topsoil (not loamy)
Z15	loamy non-calcareous sands without topsoil
Z16	slightly calcareous sands without topsoil (not loamy)
Z17	calcareous sands without topsoil (not loamy)
Z18	non-calcareous sands with clay cover (no Fe)
Z19	non-calcareous sands with clay cover, enriched with iron
Z20	calcareous sands with clay cover (no Fe)
Z30	periodically flooded sandy soils (not differentiated)
G01	gravels
L01	slightly calcareous loess
L02	calcareous loess
A01	surface water
A02	urban area, industrial areas, gravel pits, etc

Table 8.5 The 6 ECOSERIES-GWT (GroundWater Table) classes and their description, with a specification of the ecologically most discriminant factor (*cf.* Runhaar, 1989), *viz.* the Spring Groundwater Level (SGL) in cm below the surface.

GWT	DESCRIPTION	SGL
0	surface water	not relevant
1	very shallow	< 30
2	shallow	< 55
3	moderately deep	35 - 65
4	deep	55 - 100
5	very deep	> 100

Table 8.6 shows all combinations of soil units with groundwater table classes with their surface area according to the nationwide database. This table contains some remarkable facts: for example, more than 8% of the 'land' surface of the Netherlands is composed of surface water or is frequently flooded, *i.e.* GWT class 0. It may also be noted that only 10% of the country has shallow water tables, *i.e.* GWT classes 1 and 2, as a consequence of the fact that the majority of the country is intensively drained for agricultural practices (GWT classes 3 to 5).
As for the relationship with the soil aspect of the ecoseries, we may notice relatively shallow groundwater tables (classes 1 and 2) in the peat soils (coded as V..), whereas the sandy and loess soils (coded as Z.. and L.. respectively) are well drained (class 4) to excessively drained (class 5).

Summarizing, we presently have a nationwide operationalization of ecoseries in terms of a database for 1 km^2 gridcells specifying for each gridcell all ecoseries types from the 186 types that are encountered in reality, as well as their surface area accurate to 0.0025 km^2. The usefulness of this operationalization will be discussed in Chapters 9 and 10.

8.5 Discussion

In this chapter, we argued an approach to ecological land classification for mapping scales between about 1: 25,000 to 1: 100,000. Arguments for a selection of classification characteristics were taken from both the theoretical framework on hierarchical relations in ecosystems as expounded in Chapter 4, and the need to develop a practical classification that could be applied in EIA for environmental policy analyses at the national level.

Table 8.6 Surface area of all ecoseries (version 2.0) as combinations of soil (rows) and groundwater table classes (columns) in hectares for the inland surface area of the Netherlands.

SOIL	GWT 0	GWT 1	GWT 2	GWT 3	GWT 4	GWT 5	total	%
V01	0	3214	887	3118	364	0	7582	0.20
V02	2817	11408	11411	35442	4141	19	65238	1.75
V03	0	0	1151	2160	0	0	3311	0.08
V04	0	6719	96749	21391	1089	0	125948	3.38
V05	0	363	7762	7992	1289	120	17525	0.47
V06	0	0	1711	1887	0	0	3598	0.09
V07	0	0	97	538	0	0	635	0.01
V08	22	5812	75667	19402	2681	300	103884	2.79
V09	0	19	297	49	0	0	365	0.00
V10	0	48	2006	6583	508	0	9144	0.24
V11	0	0	587	5762	2904	0	9252	0.24
K01	494	34	7990	97918	54112	14182	174729	4.70
K02	13	0	704	7889	3382	112	12100	0.32
K03	18	123	10638	153605	19152	479	184013	4.95
K04	501	2095	2232	81427	280688	84790	451733	12.15
K05	76	0	216	9599	31152	11408	52451	1.41
K06	16	125	2183	24038	40920	10801	78083	2.10
K07	0	168	2762	4774	687	0	8392	0.22
K08	0	1226	34968	19048	2286	280	57807	1.55
K09	9	278	12702	7010	645	0	20643	0.55
K10	0	0	0	2214	0	0	2214	0.05
K11	0	0	0	2571	0	0	2571	0.06
K12	0	0	239	13076	8709	9863	31885	0.85
K13	0	0	127	8199	40	0	8367	0.22
K30	14415	134	141	41658	1896	171	58414	1.57
Z01	8	590	16515	103032	15780	232	136157	3.66
Z02	0	0	2015	4635	3	0	6653	0.17
Z03	0	807	2019	7576	1572	385	12358	0.33
Z04	39	14	16574	199632	148517	129085	493862	13.28
Z05	0	0	1603	23105	1071	15	25793	0.69
Z06	0	9	127	5923	1949	91	8099	0.21
Z07	0	366	3484	12224	11476	4504	32053	0.86
Z08	0	0	0	1034	1918	0	2952	0.07
Z09	0	0	12	3976	2900	931	7819	0.21
Z10	59	281	2435	11853	14283	1994	30904	0.83
Z11	0	57	100	854	3448	125	4582	0.12
Z12	189	0	1569	179486	174493	187829	543567	14.62
Z13	0	6	2192	30895	25793	17081	75966	2.04
Z14	0	0	219	1901	4029	63916	70065	1.88
Z15	0	0	0	0	0	93	93	0.00
Z16	0	328	775	419	4229	3998	9749	0.26
Z17	3	0	0	422	6316	12130	18872	0.50
Z18	20	48	8323	29059	11938	481	49868	1.34
Z19	0	0	1325	7584	580	0	9489	0.25
Z20	0	0	104	1961	6517	648	9229	0.24
Z30	5089	3053	1027	17501	1090	3220	30981	0.83
G01	1	0	0	40	0	75	116	0.00
L01	0	0	0	26302	0	0	26302	0.70
L02	0	0	136	16700	780	2888	20503	0.55
A01	294392	0	574	0	0	0	294966	7.93
A02	0	0	0	305318	0	0	305318	8.21
total	318182	37323	334351	1568781	895323	562241	3716200	
%	8.56	1.00	8.99	42.21	24.09	15.12		99.97

The hierarchies of processes and patterns and the resulting considerations as to what ecosystem components to focus on at this spatial scale have been considered as a paradigm in this context. They led only to the conclusion that soil and (ground)water would probably yield the most promising classification characteristics at this scale level, because these determine the dominant operative site factors within the context of a larger climatically and physiographically determined area.
The classification characteristics were defined more specifically by scrutinizing the relationships between operative and conditioning site factors, as well as the processes resulting from environmental hazards, in order to meet the following two criteria:

- full compatibility with the existing ecotopes classification (Stevers *et al.*, 1987; Groen *et al.*, 1993; Runhaar & Udo de Haes, 1994);
- relevance for predictive modelling in the context of EIA for environmental policy analyses.

The practical operationalization of the preliminary ecoseries classification was carried out by converting the legend to the 1: 50,000 soil map of the Netherlands, thus generalizing the more than 2,000 soil legend units into 49 relevant eco-pedological units. In combination with the 6 groundwater table classes, achieved by a similar procedure, 186 relevant ecoseries types were distinguished for the 2.0 version.

In contrast to the classifications of ecoregions and ecodistricts, the classifications of both ecotopes and ecoseries were set up in a very formal and systematic way. Two reasons can be mentioned for this, *viz.* that the classifications of ecotopes and ecoseries were developed primarily for use in effect prediction and as 'typifications' (*sensu* Zonneveld, 1994) rather than as legends to maps.
As to the first reason, we recall that effect prediction requires a thorough analysis of the relationships between the final response variable, *i.c.* the vegetation, and its abiotic environment, as well as of the abiotic processes following from human disturbance. Such an analysis yields a number of relevant classification characteristics and classes, which by systematic permutation provide a first approximation to a classification; this is consequently quite systematic.
Secondly, as for the classifications being somehow 'typifications', we wish to emphasize that both ecotopes and ecoseries are small spatial units occurring in an intricate mosaic of many thousands to millions of map units for the entire Netherlands. This requires a well-wrought typification rather than a 'holistic' map, as mapping the whole country anew is an impossible enterprise for one person. This implies that an unambiguous diagnostic instrument is required to allow easy identification by others, both in the field, or on the basis of various existing samples and/or maps. In fact, ecotope types can be derived from different types of vegetation relevées and from direct measurement of site factors, whereas the ecoseries classification was developed in such a way as to allow their identification from existing soil maps.

With respect to the 2.0 version of the ecoseries classification and its operationalization on the basis of existing maps, we wish to recall that classifications and generalizations of legends and maps for specific applications are simplified models of reality and hence always the result of an optimization procedure. This implies that, in principle, no classification can ever be finished, because it will continuously be subject to adjustments. Poore (1962) used the term 'successive approximation' in this context, which is very appropriate, as illustrated, for example, by the many approximations which preceded the US Soil

Taxonomy (Soil Survey Staff, 1975). The ecoseries classification presented in this chapter has been referred to as 2.0, deliberately referring to a predecessor and implying further developments.

Successive approximations may be needed because tests of the classification reveal shortcomings or because new applications require adaptations. So far, the classification has been used only for predicting the effects of water management on terrestrial ecosystems (Witte *et al.*, 1992; Beugelink *et al.*, 1992; Claessen *et al.*, 1994; see Chapter 10), and for forecasting the toxicological risks of chemicals for terrestrial foodwebs (Traas & Aldenberg, 1992). Applications in the field of nature conservation will be discussed in the next chapter, where we explain the conversion of ecoseries into potential site maps in operative terms. It was especially this last application that drove the development of a 2.1 version, including geographical data on upward seepage (Klijn *et al.*, 1996).

Finally, with this approach to ecoseries classification we hope to contribute to a further discussion about the ecological relevance of the interacting (ground)water and soil, requiring an approach from a combined eco-hydrological and eco-pedological point of view.

9 The ecological significance of the ecoseries classification[1]

9.1 Introduction

In the previous chapter, we defined ecoseries as ecological land units classified by conditioning site factors. This implies that they should have a certain predictive power as to (in order of causal sequence) the occurrence of abiotic sites in operative terms, and vegetations in terms of species composition. More precisely, we may expect predictive power as to the occurrence of ecotopes, as the classification was explicitly developed in relation to the existing ecotope classification (Stevers *et al.*, 1987; Groen *et al.*, 1993; Runhaar *et al.*, 1994). This may also be understood as follows: ecoseries contain ecotopes as ecotopes contain species. The reason for this is the hierarchical paradigm, explained in Chapter 4, implying that ecotopes are spatially nested within ecoseries.

The supposed predictive power of ecoseries would allow an interpretation of the database on ecoseries of the Netherlands in terms of the abiotic site types to be expected. These abiotic sites support vegetations with different vegetation structures, but all belonging to what in vegetation science is often called a sere (Tansley, 1939), or, again!, a series (Rey and Gaussen, 1955; Blasco, 1988). Thus, xeroseries can be recognized on dry sites, hydroseries on wet sites, or haloseries on saline sites.
For this interpretation of ecoseries in terms of abiotic site types we need a conversion key. This consists of so-called site diagrams (*cf.* Figure 8.3) for each ecoseries type. The concept of site diagrams will be treated in Section 9.2

9.1.1 Potential site mapping for nature conservation

With these site diagrams we can convert the nation-wide database on ecoseries into '*abiotic site maps*' as defined by operative site factors. These maps can be usefully applied for nature conservation and/or 'nature development' questions, because they can be regarded as a first step towards assessing the potential, or recovery potential, of locations in terms of vegetations to be expected.
In our approach we deviate to some extent from the usual potential vegetation mapping (after Daubenmire, 1952), which has a long tradition of focusing on which climax forest to expect at various locations (*e.g.* Tomaselli, 1973; Kalkhoven *et al.*, 1976; Conseil de l'Europe, 1987).
The first difference lies in the recognition of changing vegetation composition and structure on one and the same site. Instead of one climax forest, we should take into account various natural succession stages (*e.g.* Pfister, 1975), for example heathland or forest, as well as multiple long-lived dynamic equilibria on the same site, for example,

[1]This chapter is an extension and revision of part of Klijn, F., C.L.G. Groen & J.P.M. Witte, 1996. Ecoseries for potential site mapping, an example from the Netherlands. *Landscape and Urban Planning* 35(1996): 53-70.

closed beech forests without understorey versus oak forest with abundant understorey. Consequently, we are concerned about abiotic site factors only as a prerequisite to any natural vegetation development.

The second difference lies in a more probabilistic approach as opposed to a deterministic prediction. The interpretation of soil characteristics to yield ecologically relevant but often perishable operative site factors is not unambiguous, because of the internal heterogeneity of mapping units and the susceptibility of operative site factors to change. Instead of relating one operative site type to a mapping unit, we specify a range of site types with their estimated frequency distribution.

The possible use of the abiotic site maps will be discussed in the last section of this chapter.

9.1.2 Evaluating the ecoseries classification

Both the site diagrams and the abiotic site maps based on these may also be used for some preliminary tests concerning the ecological relevance of the ecoseries classification. This will be the core subject of this chapter, as the ecological relevance of the ecoseries is essential for any (further) practical applications.

For testing the ecoseries classification in general and its ecological relevance in particular, three aspects would need evaluation:

- firstly, it may be questioned whether the classification and operationalization of the ecoseries by means of generalizing the soil map and combining it with other data is adequate, and not either unnecessarily complex (too many types and therefore unpractical) or too generalized (too few types with an unacceptable loss of information);
- secondly, the generalization as such may be evaluated with respect to the loss of relevant distinctions and the internal homogeneity of the units;
- thirdly, the ecological significance of the resulting ecoseries types may be tested by comparing the estimated site diagrams with direct field measurements of operative site factors or indications of these site factors by plant species occurrence.

With respect to the first aspect, we cannot 'test' this, since it depends partly on the intended applications of the classification, especially the required detail for policy analyses. In this respect, the proof of the pudding is in the eating. Besides, it is a matter of opinion whether one is prepared to take into account all possibly relevant soil characteristics (resulting in over 2000 soil legend units in the 1: 50,000 soil map of the Netherlands), or merely the ecologically most discriminating ones for a nationwide analysis (49 ecoseries soils in version 2.0), or something in between (for example, the 200 units of the LKN soil classification).

As to the second aspect, the generalization as such, the method of trial-and-error may be followed to achieve the best match. The best match implies that the internal homogeneity of the legend units should be as high as possible with respect to ecologically relevant characteristics, whereas ecologically relevant distinctions in the basic maps — in our case the 1: 50,000 soil map — should be retained.

For the LKN soil generalization, De Waal (1992) compared the internal heterogeneity of the generalized units (about 200) with that of the original soil map units (over 2000) for a number of soil characteristics which were considered ecologically relevant using data on soil samples from the Soil Information System (BIS database). Texture, organic matter content, Ca-content and pH were taken into account. Unfortunately, there were insufficient data on CEC, BS, or N-total. De Waal (1992) found that all frequency diagrams showed distinct distributions, but with large variations in some units (see also Klijn & De Waal, 1992). On the other hand, this was the case in both the LKN units and the original soil map units, thus indicating no significant decline in accuracy. Obviously, the original soil map units were already quite heterogeneous in themselves.
Also the practical operationalization of the ecoseries by generalizing LKN soil units to ecoseries soils has undergone some trial-and-error, which resulted in successive approximations of the ecoseries classification and its operationalizations numbered 2.0 (Klijn *et al.*, 1992) and 2.1 (Klijn *et al.*, 1996). However, no systematic analysis of the homogeneity of ecoseries has been carried out because of insufficient data availability.

As to the third point, the evaluation of the ecological significance of the ecoseries, this has been carried out more systematically by evaluating both individual site diagrams and the 'abiotic site maps' based on these. This is the subject of Section 9.3.

9.2 Site diagrams

Ecoseries have been defined as ecological land units classified by means of conditioning site factors. However, for assessing the potential vegetation or for predicting the recovery potential they should be interpreted in terms of operative site types.
We assume that, owing to the relationship between conditioning and operative site factors on the one hand (see Section 8.3), and between operative site factors and species on the other (Runhaar & Udo de Haes, 1994), ecoseries determine ecotopes similarly to the way ecotopes determine species composition. More specifically, we may expect a characteristic frequency of abiotic sites, in operative terms, to occur in each ecoseries type.

This expectation is made explicit by specifying the supposed frequency distribution of operative site types in so-called *site diagrams* for all ecoseries types. These have the format of the general site diagram of Figure 8.3.
The site diagrams are based on the expert judgement of a team of five vegetation ecologists and soil scientists, who are well acquainted with both the ecotopes classification and the soil map, and who also have adequate knowledge of field circumstances in large parts of the country. For each type, consensus was required on a semi-quantitative estimate of the probable abundance of all operative site types. This estimate is valid for *undisturbed* conditions only, but for the *present* groundwater table class, which is usually influenced by artificial drainage.

The site diagrams are complementary to the ecoseries *classification*, but they are an indispensable complement, as they provide the link to operative site types. Essentially, they are a descriptive feature.

Legend

-:	very unlikely/ not present
+:	likely/ < 1%
1:	1-4%
2:	5-24%
3:	25-49%
4:	50-75
5:	> 75%

Primarily oligotrophic peat, not mineralized (V01) with very shallow (1, left) and shallow (2, right) groundwater table

3	+	-	+	+	-	-
4	+	-	+	+	-	-
-	-	-	-	-	-	-
-	-	-	-	-	-	-

1	+	-	+	+	-	-
4	+	-	+	+	-	-
2	+	-	+	+	-	-
-	-	-	-	-	-	-

Primarily meso-eutrophic peat enriched with iron (V06) with shallow (2, left) and moderately deep (3, right) groundwater table

+	+	-	2	1	-	-
+	2	-	4	1	-	-
+	1	-	1	+	-	-
-	-	-	-	-	-	-

-	+	-	1	1	-	-
+	+	-	2	1	-	-
+	1	-	4	2	-	-
-	-	-	-	-	-	-

Calcareous light clays and loam (not on sand or peat) (K04) with moderately deep (3, left) and deep (4, right) groundwater table

-	-	-	1	1	+	-
-	-	-	1	1	+	-
-	-	1	5	2	+	-
-	-	-	-	-	-	-

-	-	-	+	1	+	-
-	-	-	+	+	+	-
-	-	1	5	2	+	-
-	-	-	-	-	-	-

Calcareous sands without topsoil (Z17) with deep (4, left) and very deep (5, right) groundwater table

-	-	-	-	-	-	-
-	+	+	+	-	-	-
-	2	1	1	-	+	-
-	3	3	1	-	+	-

-	-	-	-	-	-	-
-	+	+	+	-	-	-
-	+	+	+	-	+	-
-	3	3	1	-	+	-

Figure 9.1 *(opposite page).*
Eight examples of site diagrams, showing the expected frequency (estimates) of operative abiotic sites for a number of ecoseries in classes.

In Figure 9.1 we give some examples for various ecoseries types (of the 2.0 approximation), namely:

- two examples (V01.1 and V01.2) of oligotrophic peat areas that are characteristic for ombrotrophic bogs, found solely in nature reserves;
- two examples (V06.2 and V06.3) of meso-eutrophic peat areas that are often influenced by upward seepage of lithotrophic groundwater, found especially in brook valleys and along the edge of the Pleistocene sandy region;
- two examples (K04.3 and K04.4) concerning fluvial and marine clay soils, mainly well-drained, as found in the best agricultural polder areas; and
- two examples (Z17.4 and Z17.5) of calcareous sands, as found in the coastal dunes, with very poor moisture retention capacity, reflected by xerophytic vegetations of nutrient-poor conditions.

9.2.1 Abiotic site maps

By multiplying the estimated shares of each operative site type according to the site diagrams by data on the surface area of ecoseries in each gridcell according to the nationwide geographical database, we can calculate the surface area of operative site types under undisturbed conditions for all gridcells. To this end, we use means for the various ranges in the site diagrams and subsequently normalize the results again to 1.00 km^2.

Thus we achieve a gridcell database indicating the percentage probability of encountering abiotic operative site types. For the 2.0 version this yields data on the potential occurrence of all operative site types as specified in Figure 9.2. From this figure it appears that some operative site types are relatively rare, such as the nutrient-poor aquatic types of various acidity classes (X11, X12, X13) and all terrestrial nutrient-poor alkaline types (X23, X43 and X63). It is noticeable that they are even more rare in surface area than in presence in gridcells. This logically follows from their small surface area per gridcell. In contrast, the moderately rich moist site type X47 is the commonest, as expected for the most mediocre type. It may be found in 92% of the gridcells, covering more than 30% of the country under undisturbed conditions. Of the dry range only the acid, nutrient-poor type (X61) is very common, especially because of its high share in cells where it occurs. It is characteristic for the Pleistocene sandy part of the country.

Next, the secondary database can be used to plot maps of the probability of encountering certain abiotic site types, indicated by X (irrespective of vegetation structure) and the code of the operative site type. Plates 9.1 and 9.2 (pages 116 and 117) are such abiotic site maps of, respectively, a common site type (X27, *i.e.* moderately rich, wet) characteristic for (lowland) peats and brook valleys, and a rare one (X63, *i.e.* nutrient-poor, alkaline, dry) characteristic for the coastal dunes and fluvial sands (of levees).

X11 5.0 % 0.06 % 1.1	X12 5.8 % 0.03 % 0.5	X13 0.2 % 0.0003 % 0.2	X17 54.6 % 5.0 % 2.5	X18 50.2 % 1.3 % 2.2
X21 39.4 % 1.2 % 2.5	X22 41.1 % 1.3 % 2.6	X23 2.4 % 0.04 % 1.5	X27 54.4 % 5.6 % 8.5	X28 45.4 % 2.5 % 4.7
X41 50.2 % 15.3 % 25.4	X42 51.5 % 4.8 % 7.9	X43 30.5 % 1.1 % 3.0	X47 91.6 % 30.5 % 11.4	X48 57.9 % 11.4 % 16.5
X61 45.8 % 21.5 % 33.7	X62 30.9 % 1.1 % 2.9	X63 3.6 % 0.5 % 11.9	X67 48.2 % 3.0 % 5.2	X68 0 % 0 % -

Figure 9.2 Potential occurrence of operative site types in the Netherlands according to the translation from ecoseries, expressed as (1) percentage of gridcells (upper, with N = 36,288), (2) percentage surface area of the inland non-urban surface (including small waters, excluding offshore areas and large water bodies) (middle; with total surface 30,265 km^2), and (3) mean surface area (lower) in hectares for cells *with* the site type.

9.3 Evaluating the site diagrams and abiotic site maps

The fact that the site diagrams and the derived maps on the potential distribution of abiotic site types are valid only for undisturbed circumstances complicates the evaluation of the ecoseries classification and operationalization. After all, the main question is whether the conversion of ecoseries into abiotic sites in operative terms is adequate.

This question can be approached by comparing the operative site conditions as *predicted* (estimated) from ecoseries, on the one hand, with the operative site conditions *indicated* by plant species occurrence (found), on the other.

Such a comparison may be carried out for individual ecoseries types with accompanying site diagrams by using vegetation relevés from known locations in nature reserves, *i.e.* in relatively undisturbed situations. Such a comparison, covering only a small number of 1: 50,000 soil map units will first be discussed (Section 9.3.1).

As the small number of available relevés and their ill representativeness hamper a thorough assessment, however, we also compare the distribution patterns of the potential site maps ('X maps') as *predicted* from the nationwide ecoseries database, on the one hand, with actual site maps, as *indicated* by plant species occurrence according to a nationwide floristic database, on the other (Section 9.3.2).

9.3.1 Reliability of individual site diagrams

The evaluation of the individual site diagrams is based on the assumption that the plant species composition of the vegetation is an accurate indication of the present site conditions. The assumption relies on the *ecological species groups*, as recognized in the ecotope classification (Runhaar *et al.*, 1987). These species groups have been tested as to both their consistency and the correlation between their occurrence and independently measured abiotic site factors (Runhaar, 1989).

Firstly, we gathered and selected existing vegetation relevés from presumably undisturbed areas (nature reserves and forests), in order to avoid large deviations from the 'virgin' state with respect to nutrient availability, groundwater regime, or soil acidity due to, for example, agricultural practice. For similar reasons we excluded vegetation relevés from those linear landscape elements that had probably been affected by fertilization or other slowly reversible human influences.
Thus, we selected 825 vegetation relevés meeting our criteria from the provinces of Gelderland (N= 233, mainly forests, small woodlands and heathlands on Pleistocene sandy soils and old, deeply weathered clay soils), Noord-Brabant (N= 115, woodlands and heathlands, predominantly on Pleistocene sandy soils), Zuid-Holland (N= 341, mainly dug-out lowland peat areas in the nature reserve of the '*Nieuwkoopse Plassen*') and Noord-Holland (N= 136, from the nature reserve '*De Zilk*' in the coastal dunes, *i.e.* on primarily calcareous sandy soils).We attempted to obtain a stratified sample of the variety in physiography and soils in the Netherlands by approximating a transect from east to west, thus encompassing the undulating Pleistocene sands (ecoregion P), the Holocene alluvial plains with clay and peat (ecoregion H) and the Coastal Dunes (ecoregion D). Unfortunately we were confronted with the complicating fact that available vegetation relevés originated mainly from the Pleistocene sandy region and dug-out lowland peats, where the majority of the nature reserves and forests in the Netherlands are found. This partially frustrated the attempt to obtain a stratified sample that was large enough to be representative for the supposedly discriminating physiographic variety.

With the computer programme ECOTYP (Stevers *et al.*, 1987) we transformed the vegetation relevés into ecotope types. Then the ecotopes were lumped into abiotic site types. Relevés indicating more than two abiotic site types were excluded. Those indicating two were divided over the respective types.
The ecoseries type (version 2.0) on which each relevé was located, was determined from the 1:50.000 soil map. Only ecoseries types with more than 10 relevés were taken into consideration in the comparison.

Thus, we could draw up site diagrams with found frequency distributions for 9 out of the 186 relevant ecoseries types. Because of the small number of available data, we transformed the found frequencies into the same classes as used for the estimates (legend to Figure 9.1). Figure 9.3 gives the estimated (left) and found (right) frequency distributions for the evaluated ecoseries types.

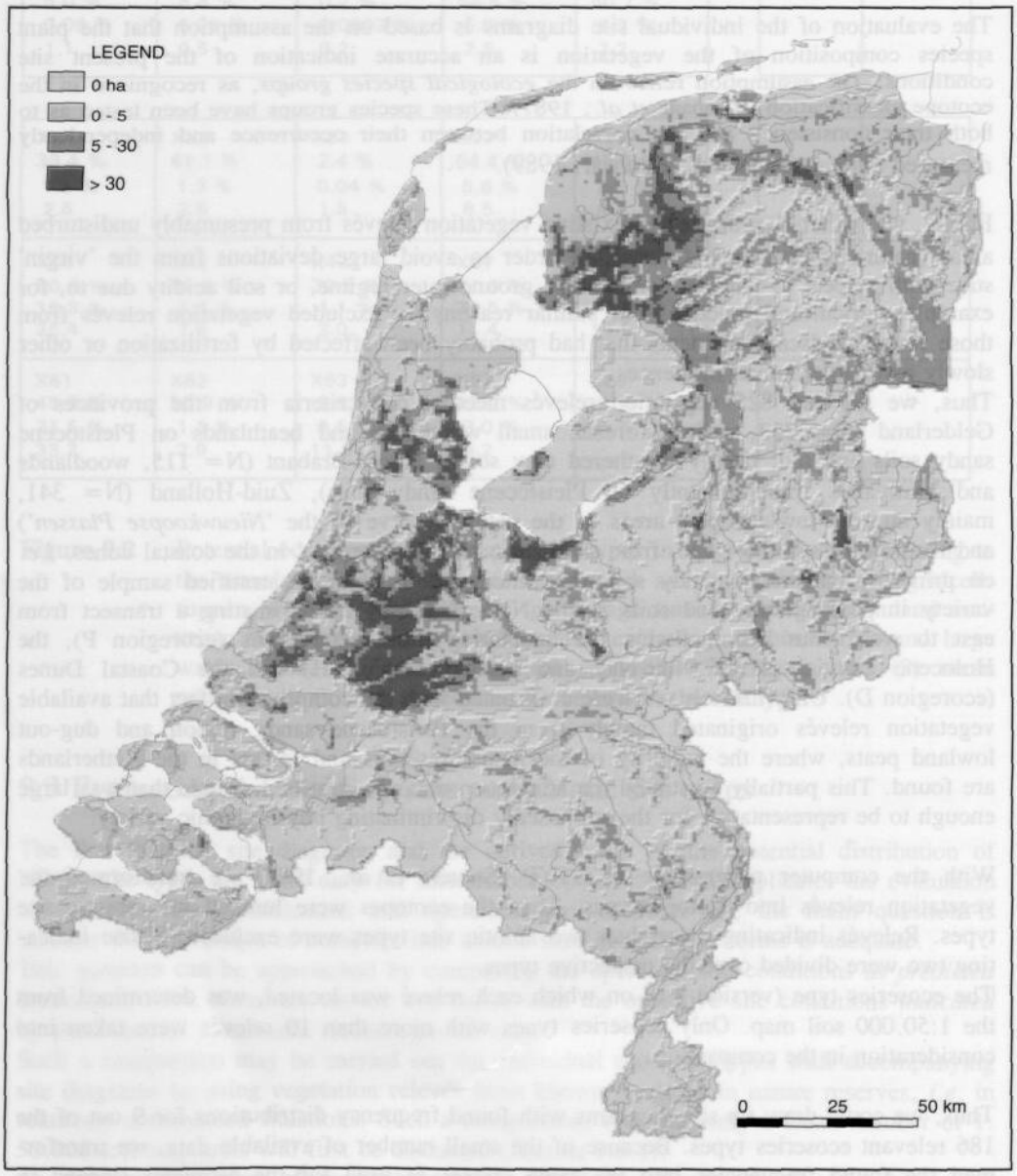

Plate 9.1 The estimated *potential* occurrence in percentage surface area per gridcell of wet, moderately rich sites (X27), as derived from the ecoseries 2.0 database. This site type is found mainly in the lowland peats and in brook valleys with upward seepage. The moderately rich type is the most common one of the wet site types.

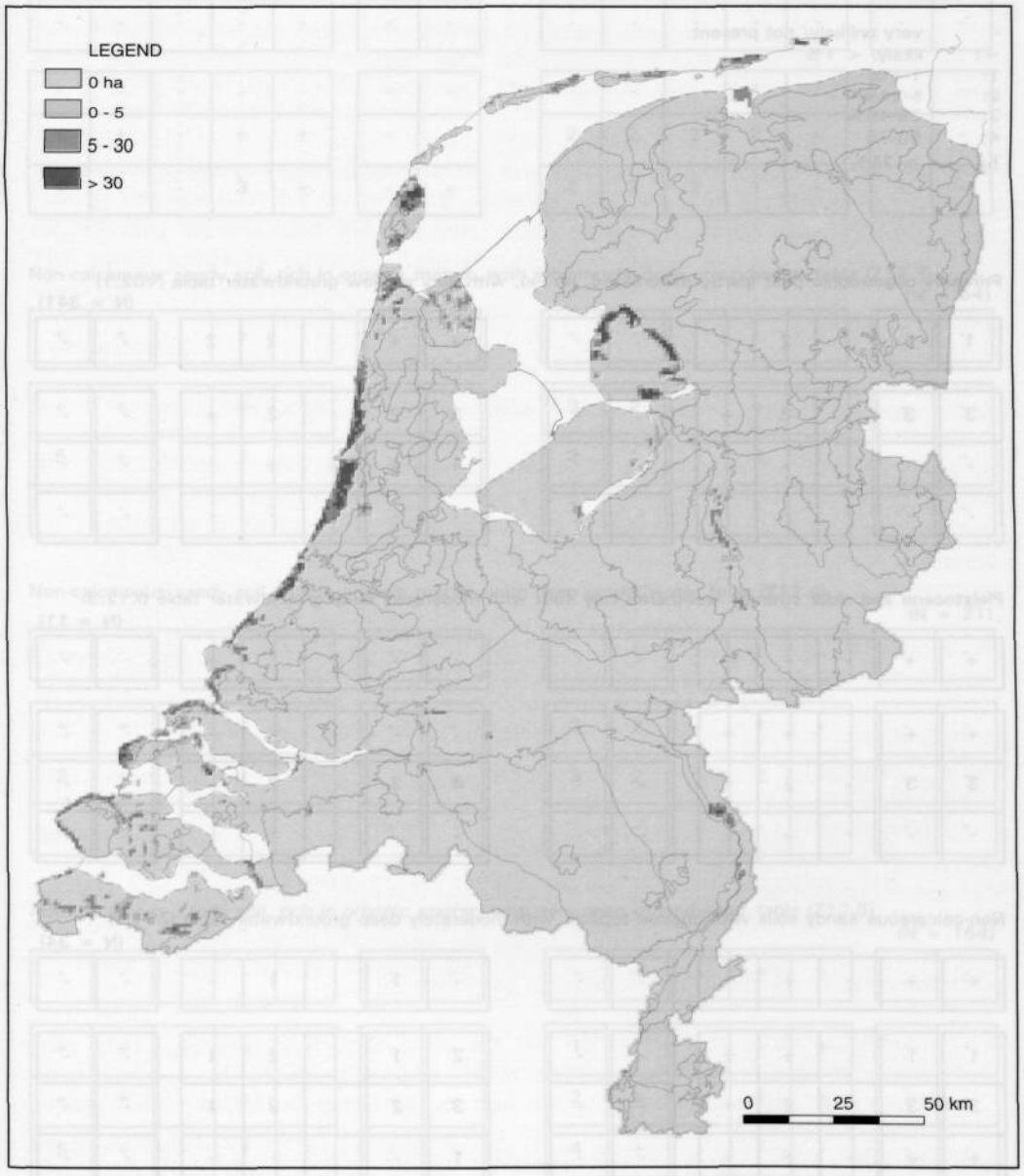

Plate 9.2 The estimated *potential* occurrence in percentage surface area of dry, nutrient-poor, alkaline sites (X63), as derived from the ecoseries 2.0 database. This site type is found mainly along the coast, especially in the coastal dunes, and on sandy natural levees in floodplain areas.

Legend to site diagrams: estimated (expected)/ found (indicated) frequency in classes

-: very unlikely/ not present
+: likely/ < 1%
1: 1-4%
2: 5-24%
3: 25-49%
4: 50-75
5: > 75%

Primarily oligotrophic peat (partly mineralized, no Fe), with very shallow groundwater table (V02.1)

(N = 341)

1	1	-	2	+	-	-	+	+	-	2	2	-	-
3	3	-	3	+	-	-	3	2	-	2	+	-	-
-	-	-	-	-	-	-	+	+	-	+	-	-	-
-	-	-	-	-	-	-	-	-	-	-	-	-	-

Pleistocene and older strongly weathered clay soils with moderately deep groundwater table (K12.3)

(N = 11)

+	+	-	+	-	-	-	-	-	-	-	-	-	-
+	+	-	+	+	-	-	-	-	-	-	-	-	-
3	3	-	2	+	-	-	4	3	-	2	-	-	-
-	-	-	-	-	-	-	-	-	-	-	-	-	-

Non-calcareous sandy soils with organic topsoil, with moderately deep groundwater table (Z04.3)

(N = 34)

+	+	-	+	+	-	-	-	1	-	1	-	-	-
1	1	-	+	+	-	-	2	1	-	2	1	-	-
3	3	-	2	+	-	-	3	2	-	2	1	-	-
+	+	-	+	+	-	-	1	-	-	-	-	-	-

Figure 9.3 Comparison of estimated frequency of abiotic sites (predicted; left diagram) with found frequencies (indicated; right diagram) for 9 ecoseries types (N = number of vegetation relevés converted to ecotope types). *Continued on next pages.*

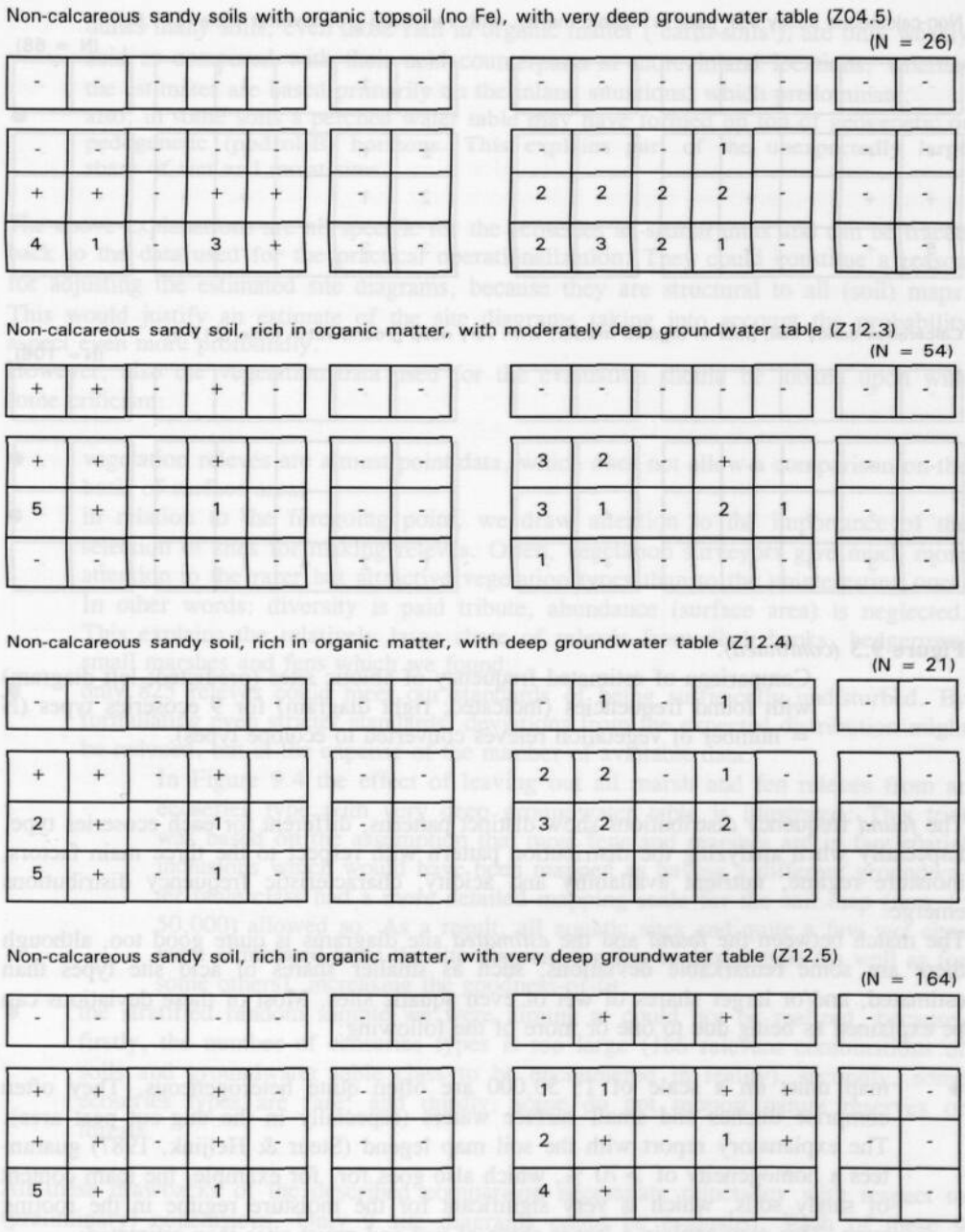

Figure 9.3 *(continued)*
Comparison of estimated frequency of abiotic sites (predicted; left diagram) with found frequencies (indicated; right diagram) for 9 ecoseries types (N = number of vegetation relevés converted to ecotope types).

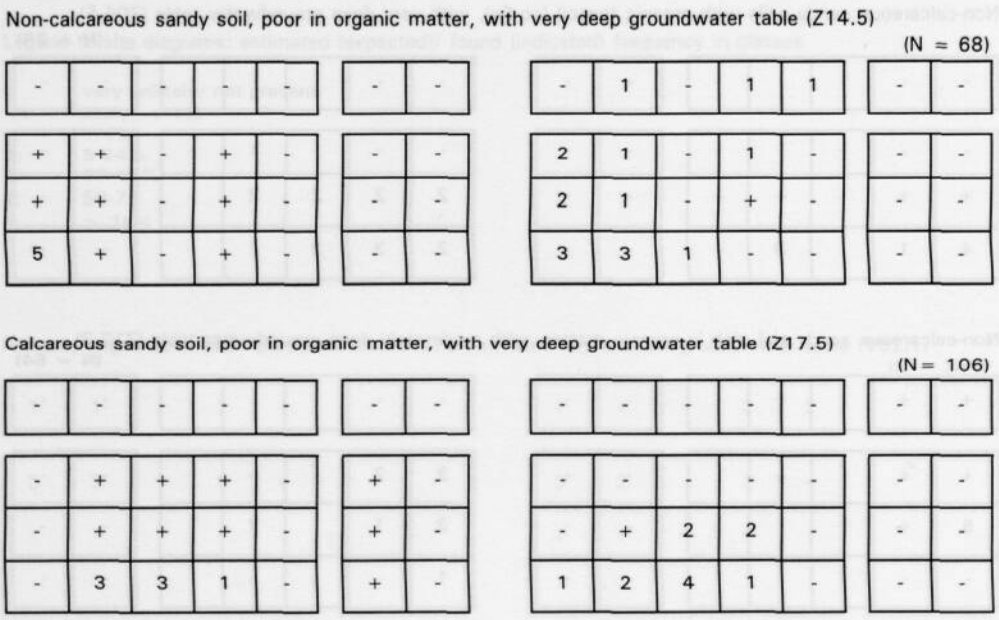

Figure 9.3 *(continued).*
Comparison of estimated frequency of abiotic sites (predicted; left diagram) with found frequencies (indicated; right diagram) for 9 ecoseries types (N = number of vegetation relevés converted to ecotope types).

The *found* frequency distributions show distinct patterns, different for each ecoseries type. Especially when analyzing the distribution pattern with respect to the three main factors, moisture regime, nutrient availabiliy and acidity, characteristic frequency distributions emerge.
The match between the *found* and the *estimated* site diagrams is quite good too, although there are some remarkable deviations, such as smaller shares of acid site types than estimated, and/or larger shares of wet or even aquatic sites. Most of these deviations can be explained as being due to one or more of the following:

- map units on a scale of 1: 50,000 are often quite heterogeneous. They often comprise ditches and small surface waters (especially in the dug-out peat area). The explanatory report with the soil map legend (Steur & Heijink, 1987) guarantees a homogeneity of >70 %, which also goes for, for example, the loam content of sandy soils, which is very significant for the moisture regime in the rooting zone. This may explain a larger share of wet and moist sites than estimated;
- some legend units are characterized by considerable micro-relief, especially in dune areas, with corresponding heterogeneity in groundwater table classes and weakly developed but characteristic soil catenas;
- hardly noticeable differences in $CaCO_3$-content which are reflected in species composition are not distinguished on the soil map. For example, in the coastal

dunes many soils, even those rich in organic matter ('earth-soils'), are only weakly acid as compared with their acid counterparts at more inland locations, whereas the estimates are based primarily on the inland situations, which predominate;

- also, in some soils a perched water table may have formed on top of geogenetic or pedogenetic (podzol-B) horizons. This explains part of the unexpectedly large share of wet and moist sites.

The above explanations are all specific for the ecoseries as spatial units and can be traced back to the data used for the practical operationalization. They could constitue a reason for adjusting the estimated site diagrams, because they are structural to all (soil) maps. This would justify an estimate of the site diagrams taking into account the probability aspect even more profoundly.
However, also the vegetation data used for the evaluation should be looked upon with some criticism:

- vegetation relevés are almost point data, which does not allow a comparison on the basis of surface area;
- in relation to the foregoing point, we draw attention to the importance of the selection of sites for making relevés. Often, vegetation surveyors give much more attention to the rarer but attractive vegetation types than to the uninteresting ones. In other words: diversity is paid tribute, abundance (surface area) is neglected. This explains the relatively large share of relevés from ditch banks, hedgerows, small marshes and fens which we found;
- only 825 relevés could meet our standards of being sufficiently undisturbed. By formulating even stricter standards, deviations from the expected distribution might be reduced, but at the expense of the number of available data.

  In Figure 9.4 the effect of leaving out all marsh and fen relevés from an ecoseries type with very deep groundwater table is illustrated. This trial was based on the assumption that most fens and marshes are in fact spatial inclusions which would have been mapped as having a different groundwater table class had a more detailed mapping scale for the soil map (now 1: 50,000) allowed so. As a result, all aquatic sites and quite a few wet sites were eliminated from the site diagram for this ecoseries type (as well as for some others), increasing the goodness-of-fit;
- the stratified random sample we were aiming at could not be realized, because, firstly, the number of ecoseries types is too large (186 relevant combinations of soils and groundwater table class to be encountered in reality), secondly, some ecoseries types are rare, and thirdly, some do not support nature reserves or forests.

All these drawbacks of the described comparison necessitate reluctancy with respect to (preliminary) conclusions. Only 9 site diagrams could be evaluated. Each of these 9 shows a characteristic frequency distribution of operative site types, which differs from those of the other evaluated types. Whether this difference is significant has not been tested, as it requires more data and coverage of all ecoseries types.
The estimated site distribution largely resembles the distribution found in the field surveys. A formal test as to their similarity cannot be carried out because of the interdependence of the factors involved. The number and quality of the data used for the

comparison do not justify adjustments to the estimated site diagrams of version 2.0, however. This means that we must continue our efforts to evaluate the ecological significance of the present version of the ecoseries classification and the related site diagrams.

Legend to site diagrams: found frequency (classes)

-:	not present
+:	< 1%
1:	1-4%
2:	5-24%
3:	25-49%
4:	50-75
5:	> 75%

Non-calcareous sandy soil, poor in organic matter, with very deep groundwater table (Z14.5)

N = 68							N = 61						
-	1	-	1	1	-	-	-	-	-	-	-	-	-
2	1	-	1	-	-	-	1	+	-	+	-	-	-
2	1	-	+	-	-	-	2	1	-	+	-	-	-
3	3	1	-	-	-	-	3	3	1	-	-	-	-

Figure 9.4 Differences in found (indicated) frequency of abiotic sites for one of the ecoseries from Figure 9.3 *with* (left) and *without* (right) vegetation relevés from fens and marshes.

9.3.2 Comparing potential site distribution with actual site distribution

Another possible test consists of a comparison of the *predicted* (potential) distribution of the abiotic site types with their *indicated* (actual) occurrence. This requires access to nationwide gridcell databases of equal resolution.
In the previous chapter we already introduced the nationwide operationalization of the ecoseries classification on a 1 km^2 grid, which is tuned to the national 'Amersfoort grid'. The same grid is used for both a floristic database: FLORBASE (Groen *et al.*, 1992) and a vegetation and land-cover database compiled for the Landscape Ecological Mapping of the Netherlands (Van der Linden *et al.*, 1995). Consequently, there are no technical obstacles to such a comparison.
As, however, the majority of the cells is (at least partly) disturbed, *e.g.* agricultural or urban area, the evaluation may be carried out in only one direction: is the abiotic site type predicted when it is actually indicated by plant species?

For establishing the abiotic site conditions from the occurrence of plant species or vegetation we again used the indicative value of *ecological species groups* (Runhaar *et*

al., 1987). These species groups allow the entire species composition of a relevé to be used to define the abiotic site type (*cf.* Witte & Van der Meijden, 1990). Similarly, all the species found in a 1 km^2 gridcell can be used to determine which abiotic site types (predominantly plural) occur in that specific cell. To this end, we again presume the species to be good indicators of actual site conditions (Runhaar, 1989).
Thus, from FLORBASE (version 0) we determined the geographical distribution of about 30 aquatic, woody and herbaceous vegetations of a number of abiotic site types (see Plates 9.3 and 9.4 on pages 124 and 125). A total of 23,816 gridcells were considered sufficiently sampled for this exercise, whereas 9,211 passed an even stricter selection.
Also from the LKN database on vegetation and land cover we determined the geographical distribution of a number of abiotic site types.

Taking into consideration that the data on flora and vegetation are incomplete (*cf.* Groen *et al.*, 1992 and Van der Linden *et al.*, 1995), Plates 9.1 and 9.3 show remarkable similarities in distribution pattern, as do Plates 9.2 and 9.4. It seems that the indicated and predicted distributions of site types correspond quite well.

Due to the same incompleteness of the data on flora and vegetation, we can only calculate to what extent *indicated* occurrence overlaps *predicted* occurrence, using the following formula, weighted to completeness 'C' (Witte & Van der Meijden, 1992):

$$\text{Overlap} = \frac{\Sigma C \text{ (cells with ecotope on site)}}{\Sigma C \text{ (cells with ecotope)}}$$

The overlap with the distribution according to FLORBASE-0 is expressed in Figure 9.5, and with that according to the LKN vegetation data in Figure 9.6.
The overlap percentages specified in these figures are remarkably high. However, for abiotic site types that are found in almost all the gridcells (*e.g.* X47, which is predicted in 92% of the gridcells, see Figure 9.2) high overlaps are not significant; a suitable site type is nearly always present. Therefore, we also calculated a factor to indicate to what extent the overlap exceeds 'random overlap', using the following formula (Witte & Van der Meijden, 1992):

$$\text{Factor} = \frac{\text{Found overlap}}{\text{Expected overlap (random)}}$$

implying that the factor is 1.0 at random overlap. This factor can be regarded as a kind of significance factor, giving additional information to the mere significance test with X^2 (Siegel & Castellan, 1988).

From Figures 9.5 and 9.6 we may conclude that relatively extreme site types, such as dry, poor calcareous (X63) or wet, poor, weakly acid (X22), can be predicted relatively accurately. Also, for these site types distinct indicative species groups exist. In practice, such extreme site types are also very rare. In contrast, for the most common site type to be expected, *i.e.* X47, the overlap fraction is very high indeed, but the factor is a mere 1.0, which is not significant.

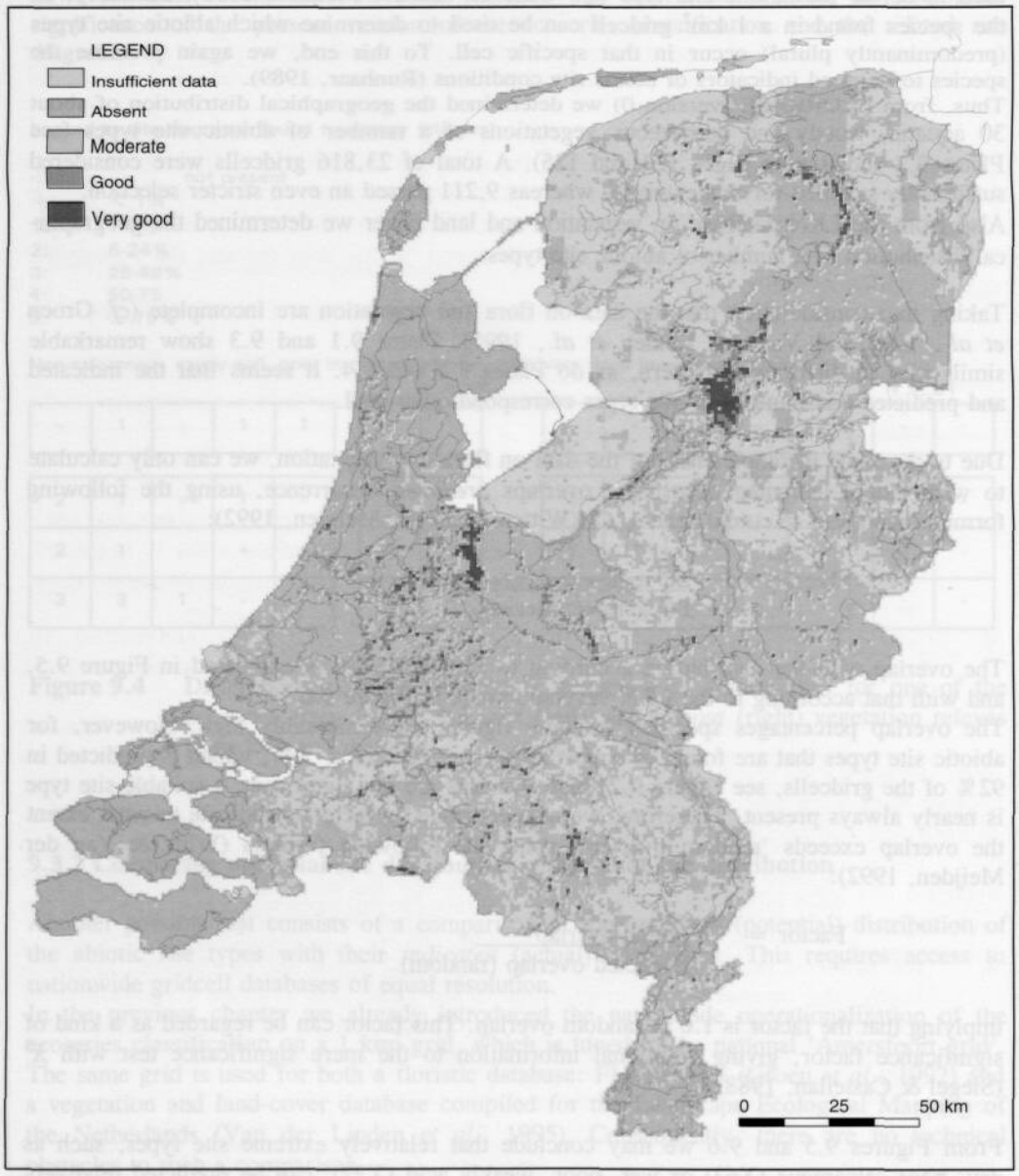

Plate 9.3 The *actual* occurrence of wet, moderately rich sites (X27), as indicated by plant species occurrence according to the floristic database FLORBASE expressed in four classes of completeness relative to type-specific lower and upper thresholds (Witte & Van der Meijden, 1992).

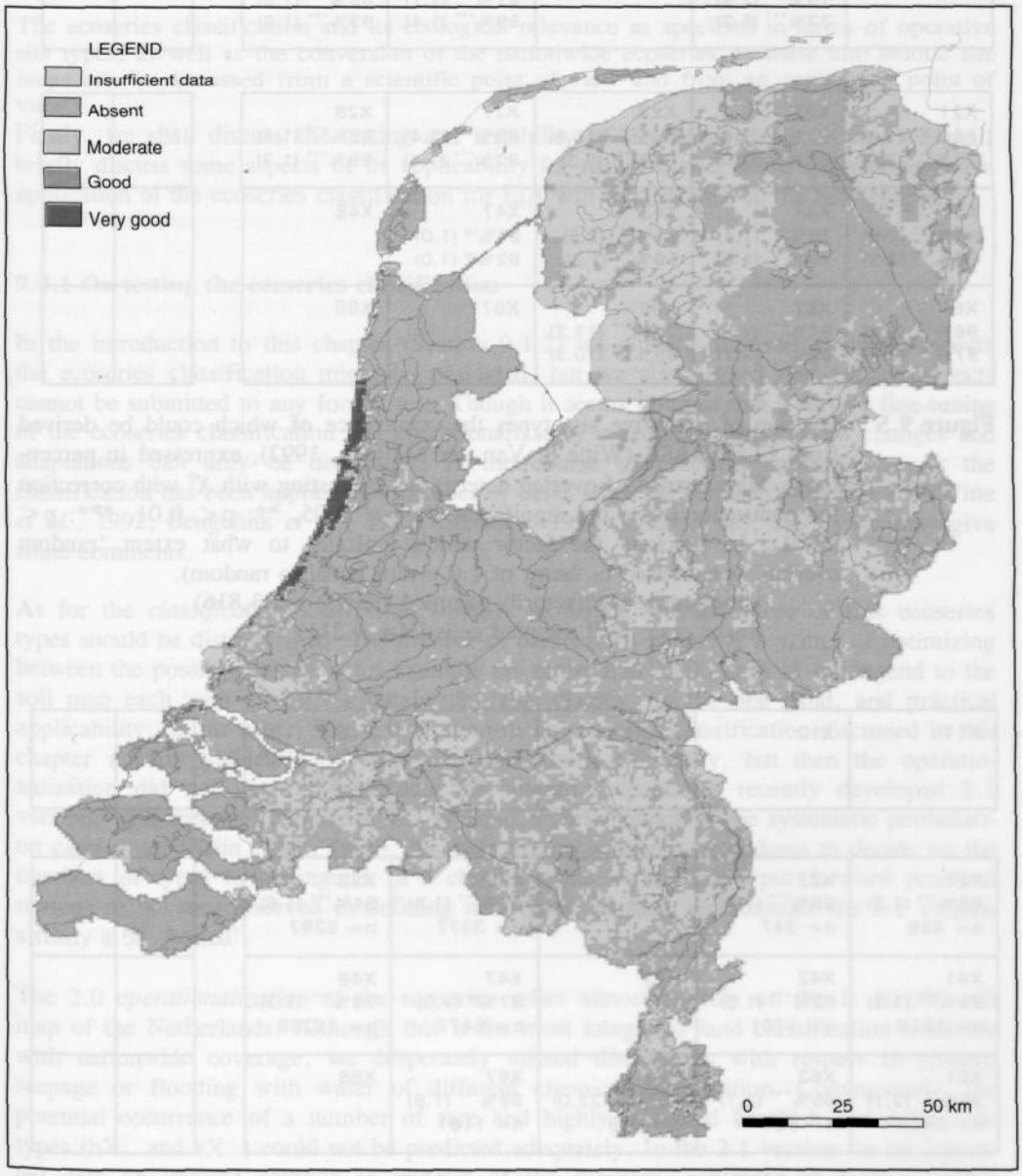

Plate 9.4 The *actual* occurrence of dry, nutrient-poor, alkaline sites (X63), as indicated by plant species occurrence according to the floristic database FLORBASE expressed in four classes of completeness relative to type-specific lower and upper thresholds (Witte & Van der Meijden, 1992).

X11	X12 15%*** (2.4) 32%*** (5.2)	X13	X17 97%*** (1.7) 99%*** (1.4)	X18 85%*** (1.5) 93%*** (1.2)		
X21 68%*** (1.6) 73%*** (2.0)	X22 87%*** (2.0) 89%*** (2.0)	X23 64%*** (16.8) 65%*** (9.3)	X27 83%*** (1.4) 92%*** (1.3)	X28 83%*** (1.6) 89%*** (1.3)		
X41 95%*** (1.9) 96%*** (2.9)	X42 76%*** (1.4) 73%*** (1.6)	X43 81%*** (2.5) 59%** (1.3)	X47 94%ns (1.0) 92%ns (1.0)	X48		
X61 96%*** (2.2) 97%*** (3.5)	X62 61%*** (1.9) 63%*** (2.7)	X63 77%*** (17.2) 85%*** (10.3)	X67	X68		

Figure 9.5 Overlap of operative site types the occurrence of which could be derived from FLORBASE (Witte & Van der Meijden, 1992), expressed in percentages. Significance of overlap determined by testing with X^2 with correction for continuity (n.s.: not significant, *: $p < 0.05$, **: $p < 0.01$, ***: $p < 0.001$). In brackets the factor which indicates to what extent 'random overlap' is exceeded (a factor of 1.0 means equal to random).
upper case: satisfactorily sampled cells (N= 23,816)
lower case: well sampled cells (N= 9,211)

X11	X12	X13	X17 88%*** (1.5) n= 2682	X18 81%*** (1.5) n= 9801		
X21 66%*** (1.2) n= 436	X22 66%*** (1.6) n= 347	X23	X27 81%*** (1.4) n= 3577	X28 84%*** (1.6) n= 5297		
X41 89%*** (1.7) n= 1616	X42 82%*** (1.6) n= 1701	X43	X47 97%ns (1.0) n= 6477	X48 78%*** (1.3) n= 11269		
X61 98%*** (2.1) n= 2464	X62 65%*** (2.1) n= 621	X63 88%*** (22.0) n= 379	X67 89%*** (1.8) n= 1161	X68		

Figure 9.6 Overlap of operative site types (presence only) the actual occurrence of which could be derived from the LKN vegetation/land cover database, expressed in percentages. Significance of overlap determined by testing with X^2 corrected for continuity (n.s.: not significant, *: $p < 0.05$, **: $p < 0.01$, ***: $p < 0.001$). In brackets the factor that indicates to what extent 'random overlap' is exceeded.

9.4 Discussion

The ecoseries classification and its ecological relevance as specified in terms of operative site types, as well as the conversion of the nationwide ecoseries database into abiotic site maps can be discussed from a scientific point of view and from an application point of view.

Firstly, we shall discuss the testing and testability of the classification. Then, we shall briefly discuss some aspects of its applicability for nature conservation and recovery. An application of the ecoseries classification for EIA will be discussed in the next chapter.

9.4.1 On testing the ecoseries classification

In the introduction to this chapter (Section 9.1.2) we already discussed on what aspects the ecoseries classification might be evaluated, but we also argued that various aspects cannot be submitted to any formal test. Though it seems obvious that a further fine-tuning of the ecoseries classification and operationalization is required, the kind of changes and adaptations can only be determined in the course of practical applications. As the classification has been applied in a number of cases since the release of version 2.0 (Witte *et al.*, 1992; Beugelink *et al.*, 1992; Claessen, 1994; Traas *et al.*, 1995), we may give some comments.

As for the *classification* itself, it is difficult to decide whether more or less ecoseries types should be distinguished. The number of distinctions is always a matter of optimizing between the possibly relevant, for example the more than 2,000 units of the legend to the soil map each with up to 7 groundwater table classes, on the one hand, and practical applicability, on the other. The 2.0 version of the ecoseries classification discussed in this chapter already yielded 186 types that were found in reality, but then the operationalization did not yet comprise data on upward seepage. A recently developed 2.1 version, with seepage, comprises 432 types as a consequence of the systematic permutation carried out (Klijn *et al.*, 1996). As there are no scientific procedures to decide on the numbers of types to distinguish in a classification, we can only put forward practical reasons to be very reserved in defining new types. In fact, we consider the 2.1 version already a bit 'bolted'.

The 2.0 *operationalization* of the ecoseries relies almost entirely on the 1: 50.000 soil map of the Netherlands. Although this is the most integrated land classification available with nationwide coverage, we desperately missed distinctions with respect to upward seepage or flooding with water of different chemical composition. Consequently, the potential occurrence of a number of rare and highly esteemed brackish and saline site types (bX.. and zX..) could not be predicted adequately. In the 2.1 version we incorporated data on upward seepage in updates of the database, achieving at least a partial improvement.

In this chapter, site diagrams have been presented as an indispensable complement to the ecoseries classification, because only through these is an interpretation in terms of operative site types possible.

A formal, though partial, evaluation of the combination of the ecoseries classification, its operationalization and the accompanying site diagrams has been presented in this chapter. Despite a number of drawbacks as to the number and quality of the data in relation to the large number of ecoseries for which the site diagrams had to be evaluated, we dare conclude that the approach as such is consistent and useful. However, an evaluation of individual site diagrams could be attained for only 9 ecoseries types from a total number of 186 in the 2.0 version. As the number of ecoseries types has meanwhile increased to 432 types in version 2.1, a thorough evaluation of all site diagrams will probably remain wishful thinking, especially because for many ecoseries types it appears practically impossible to obtain sufficient data without starting a laborious sampling scheme to obtain some thousands of vegetation relevés.
Also the overlap calculations of the spatial distribution of sites presented in this chapter can only be regarded as a preliminary test for the viability of the approach, mainly because the potential site maps are evaluated, and not the individual site diagrams. This implies a test of the 'end results' of a number of subsequent steps all relying partly on expert judgement.

In this context, we would like to emphasize again that the improvement of any classification, its operationalization and its applications is a matter of successive approximation, and we ask renewed attention for what we stated earlier on this matter in Section 8.5.

9.4.2 On potential site mapping for nature conservation and recovery

With reference to the title of this section, we first want to emphasize that we realize that nature conservation and recovery projects generally focus on a more detailed scale level than the 1 km^2 gridcell. This explains why the ecoseries *classification* is related to the 1: 50,000 soil map in tune with the guidelines on spatial scales as proposed in Chapter 4. This allows an easy scale change to the geographically more confined areas usually at stake in nature conservation and recovery, and it also permits more detailed predictions concerning individual mapping units of the soil map (*cf.* Boersma, 1995). The ultimate practical limitations are then posed by the accuracy of the soil map and the reliability of the site diagrams. The *operationalization* in a 1 km^2 gridcell database, however, was intended to provide a nationwide overview for policy decisions at the national level.

The use of site diagrams may be considered an advance in attempts to estimate 'potential nature', when compared with the more traditional way of expressing this as 'potential vegetation', mostly a climax vegetation type.
A first improvement is formed by the probabilistic character of the site diagrams in comparison with the more deterministic approach of defining 'potential vegetations' in terms of climax vegetation. In fact, the classes used in the site diagrams can be read as probabilities of occurrence of certain operative sites on that particular ecoseries type.
The second improvement is the terminology, whereby the use of 'operative site types' instead of 'potential vegetation' can be considered an advance for estimating 'potential nature', leaving open the choice on how to manage vegetation structure, *e.g.* by mowing, grazing, cutting, or whatever else.

The nationwide potential abiotic site maps and the overview of rarity of each operative site type are especially important for nature conservation policy at the national level, allowing due focus of attention and means. Information of this kind can be used to prioritize between nature development projects or restoration projects as proposed by local or regional authorities or NGO's applying for subsidies, because it allows proposals to be assessed in a national and quantified context. Especially the potential rarity of an abiotic site type may be an important additional criterion in this respect, supplementary to the actual rarity of species or vegetations of certain site types. For example, the actual and potential distributions of plant species of site type X63 (Plates 9.2 and 9.4), which is potentially rare, do not deviate very much, because the coastal dunes are largely well protected and have a nature conservation function. In contrast, plant species indicative of site type X27, *i.e.* wet, moderately rich sites (Plates 9.1 and 9.3) are much more common, but the discrepancy between actual distribution and potential distribution is large (Witte & van der Meijden, 1990; 1992; Groen *et al.*, 1994). Actual rarity may thus be relativized.

10 Applying ecoseries in hydro-ecological modelling for national water management[1]

10.1 Introduction

In Chapter 8 we stated that the ecoseries classification was designed primarily for environmental impact assessment (EIA) by sustaining predictive modelling. In this context, it was developed as an interface between human interventions and/or immissions on the one hand, and biotic response due to changing site conditions on the other.
This chapter treats the use of the ecoseries classification in the predictive model DEMNAT (Dose-Effect Model for terrestrial NATure; Witte, 1990; Witte *et al.*, 1992, 1992a, 1993; Nienhuis *et al.*, 1992), illustrated by an example of its practical and effective use in an EIA for the future public water supply of the Netherlands (Beugelink *et al.*, 1992).
With DEMNAT plant species loss and gain in response to water management measures and changes in groundwater extraction can be assessed, applying both the ecoseries and ecotope classifications.

Before treating DEMNAT and related tools, we shall briefly introduce the 'desiccation' problem in the Netherlands and the policy context for which these tools have been developed.

10.1.1 Effects of water management on terrestrial nature values

Since about 1950, major interventions in the Netherlands' water management have resulted in severe damage to terrestrial ecosystems at wet and moist sites. These interventions have caused lowered groundwater levels and diminished intensity of upward seepage, as well as being accompanied by intentionally letting in surface water of poor quality to compensate for water shortages (*cf.* Duel *et al.*, 1989; Fiselier *et al.*, 1992). The combined effects on ecosystems are addressed with the term 'desiccation'[2] or drought damage, even when these effects are due not to an absolute shortage of water but to the changed water quality.

In the eighties 75 % of wet and moist sites were found to suffer from drought damage (Projectgroep Verdroging, 1989). The main interventions causing this deterioriation of

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[1]This chapter includes strongly revised texts from Claessen, F.A.M., F. Klijn, J.P.M. Witte & J.G. Nienhuis, 1994. Ecosystem classification and hydro-ecological modelling for national water management. Pages 199-222 in Klijn, F., 1994. *Ecosystem classification for environmental management*. Kluwer Academic Publishers, Dordrecht.

[2]Drought damage and 'desiccation' are both inadequate translations of the Dutch '*verdroging*'. 'Verdroging' is one of the themes for the Netherlands' environmental policy used to indicate a group of related environmental problems resulting from water shortages due to groundwater extraction and enhanced drainage.

nature values are assumed to be increased drainage of agricultural and urban areas, intensified regulation of surface water levels, doubling of groundwater extraction for the public water supply since the fifties, and increased sprinkling irrigation in agriculture using both groundwater and surface water.

These interventions have changed the hydrological conditions within and around nature conservation areas. In general, they have resulted in a lowering of the groundwater levels between 0.2 and 1.0 metre, as well as a lowering of the surface water level by between 0.1 and 0.5 metre (Rolf, 1989). Changes have also occurred in the quality of both groundwater and surface water, involving a lowering of the pH or an increase in the availability of nutrients in the rooting zone. Consequently, the occurrence of wet or moist, nutrient-poor, pH-buffered conditions is currently confined to only a few small areas.

10.1.2 Instruments for water-management policy analysis

The recognition of desiccation as an important environmental problem (Runhaar *et al.*, 1996) has resulted in a revision of ideas on water use and water management. The revised ideas have been made explicit in policy plans concerning national water management in general (Ministry of V&W, 1989) and public and industrial water supply, more in particular (Ministry of VROM, 1993).

Generally, policy analyses are carried out to provide scientific support for the policy-making process concerning surface-water and groundwater management. To this end, a first set of predictive models was developed for the Policy Analysis of the Water management of the Netherlands (PAWN; Pulles, 1985). This tool-kit for policy analysis at the national level has since been extended continuously.
In this context, a model to predict the ecological effects of water-management alternatives was developed as well; the first version of the Dose-Effect Model for terrestrial NATure (DEMNAT) (Claessen & Witte, 1989). This model is suitable for determining changes in the nature value of terrestrial ecosystems resulting from changes in abiotic conditions following water-management interventions.
Recently, DEMNAT has been adapted to enable it to be applied in an environmental impact assessment for the national Policy Plan on Public Drinking Water Supply and Industrial Water Supply (Beugelink *et al.*, 1992; 1993). This 2.0 version is presently included in the tool-kits of both the National Institute for Inland Water Management and Waste Water Treatment (RIZA) and the National Institute of Public Health and Environmental Protection (RIVM). Although another update is currently being worked on, we shall refer to this second (2.0) version of DEMNAT in this chapter, as it uses the 2.0 version of the ecoseries classification.

After this general introduction on the context of DEMNAT's development, we shall first give a short overview of the concepts and tools for a nationwide assessment of the ecological effects of water-management alternatives. Next, we briefly describe the DEMNAT model, followed by an illustration of its application in the EIA for the national Policy Plan on Public Drinking Water Supply and Industrial Water Supply. We shall end with a brief discussion with special reference to the use of our classifications.

10.2 Concepts and tools

Predictive modelling is the predominant approach to answering policy questions concerning the effects of various water-management scenarios. With regard to the *ecological* effects of water-management scenarios on terrestrial ecosystems, we limited the study to the vegetation as the prime affected component. The results of the predictive modelling had to be expressed in both objective terms, such as loss of surface area and species richness, and normative terms, *i.e.* nature value.

The *hydro-ecological* modelling was based on the following chain of processes (see Figure 10.1):

- Water-use and water-management interventions result in changes in local hydrological conditions. Partly, these are ecological key factors, especially the spring groundwater level (SGL), the intensity of upward seepage, the surface water level, and the share of surface water of poor quality let in from the 'main system'.
- The local hydrological changes result in changing soil-water behaviour, the mineralization of organic matter and soil acidification. These processes are controlled by *conditioning* factors, such as soil texture, organic matter content, $CaCO_3$-content, groundwater level and the chemical composition of seepage water.
- These processes, in turn, result in changes in *operative* site factors, namely moisture availability, nutrient availability and acidity in the rooting zone.
- Finally, the changes in operative site factors result in changes in the species composition of the vegetation. These changes are expressed in terms of relative species richness or 'completeness', and in terms of nature value.

10.2.1 Prediction models

For the various steps in this chain of effects, different models were used or newly developed. Firstly, a national groundwater model (LGM; Pastoors, 1992; 1992a) was developed to calculate the changes in groundwater levels, the changes in the water levels of small surface waters and the changes in the upward seepage of groundwater, all due to changes in groundwater withdrawal. These calculations were carried out per gridcell of 1 km^2.
Secondly, a national surface water distribution model (DM-DEMGEN) was used to estimate the changes in water quality within 80 water-management districts due to letting in polluted water from the rivers Rhine and Meuse (Pakes *et al.*, 1992). It was used to calculate the ratio between the volumes of intake water and the resident volume of surface water within a district during the growing season. The higher the share of intake water, the greater the supposed adverse effects on surface water quality.

The aforementioned two models are hydrological models, used primarily for calculating the changes in local hydrological conditions. The third model, then, is DEMNAT (Witte, 1990; Witte *et al.*, 1992a and b; Nienhuis *et al.*, 1992) intended for predicting the biotic response to the local hydrological changes. For DEMNAT (see Section 10.3), the hydrological changes at site level function as a dose, whereas the effects are expressed in terms of changed plant species composition, as well as in terms of changed nature value.

10.2.2 Ecological interfaces

Hydrological changes cannot be directly translated into changes in the species composition of the vegetation, because a number of abiotic processes in abiotic components are involved. Some of these, such as mineralization or acidification, may aggravate the effects of groundwater lowering, whereas others, such as water retention or capillary rise, may diminish the effects.

Because of these intermediate abiotic processes, we needed interfaces between the hydrological doses on the one hand and the biotic response on the other. As illustrated in Figure 10.1, we used the classifications of ecoseries and ecotopes as interfaces. The ecoseries classification relates the operative site factors to conditioning soil and groundwater characteristics, whereas, subsequently, the ecotope classification relates the species composition of the vegetation to the operative site factors. Since they are classified by conditioning factors, the ecoseries also determine the response of the operative site factors to changes in the local hydrology (see Figure 10.1, left margin). And the ecotopes determine the response of the species composition to the changes in the operative site factors (see Figure 10.1, left margin).

The combination of the two classifications yields the spatial units for which all further calculations are carried out. In mapping terms, this combination can be regarded as an overlay of an ecological vegetation map (ecotopes) and an ecological soil/groundwater map (ecoseries), where the ecotopes are nested within the ecoseries.

10.3 DEMNAT

The predictive model DEMNAT contains three essential parts, namely a geographical database of the Netherlands with respect to the spatial distribution of ecosystems, dose-effect functions, and a nature valuation system. These parts will be described successively. Because we shall refer to the 2.0 version of DEMNAT as it was used for the EIA for the national public water supply policy, a number of points made below are specific to this nationwide study, but not inherently connected with DEMNAT's structure as such. DEMNAT may also be used with other data on regional or even local scales.

10.3.1 Geographical database

One of the main problems that needed solving was how to assess the impact of changes in water management on the national scale on terrestrial ecosystems on the local scale, *i.e.* ecotope scale level. Additional complicating factors originated from the need to make predictions for the entire country, and the poor availability of accurate data on vegetation for such a large area.

The hydrological models permitted doses to be calculated with a spatial accuracy of about 1 km^2. For the distribution of intake water, this already required a scaling down of calculations concerning 80 water management districts to 1 km^2 gridcells. Consequently, a higher accuracy than 1 km^2 was regarded as unachievable. Moreover, a 1 km^2 grid would allow the whole country to be represented with only about 36,000 gridcells, which was considered just manageable.

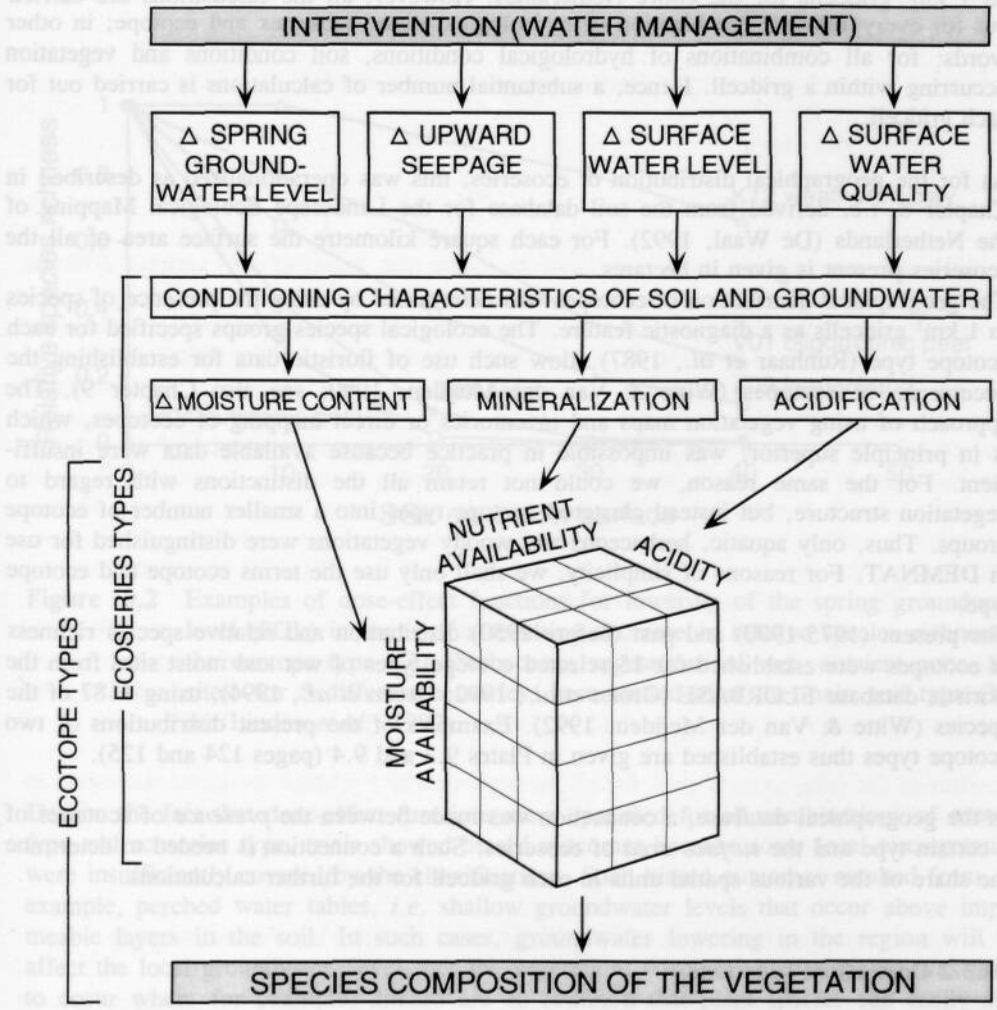

Figure 10.1 Process chain resulting from interventions in water management, distinguished for predicting the subsequent responses of operative site factors and species composition of the vegetation (adapted from Van Beusekom *et al.*, 1990). The connection with the ecoseries and ecotope classifications is indicated in the left margin.

Also data on plant species composition were available at no greater resolution than for 1 km^2 gridcells, while data on soil and groundwater levels were also readily available on this gridcell level (see Chapter 8).
Therefore, it was decided that the nationwide geographical database was to contain data on 1 km^2 gridcells for the entire Netherlands. However, all the calculations are carried out for every relevant combination of hydrological dose, ecoseries and ecotope; in other words: for all combinations of hydrological conditions, soil conditions and vegetation occurring within a gridcell. Hence, a substantial number of calculations is carried out for each gridcell.

As for the geographical distribution of ecoseries, this was operationalized as described in Chapter 8, *i.e.* derived from the soil database for the Landscape Ecological Mapping of the Netherlands (De Waal, 1992). For each square kilometre the surface area of all the ecoseries present is given in hectares.
The geographical distribution of ecotopes was determined by using the presence of species in 1 km^2 gridcells as a diagnostic feature. The ecological species groups specified for each ecotope type (Runhaar *et al.*, 1987) allow such use of floristic data for establishing the occurrence of ecotopes (Witte & Van der Meijden, 1990; see also Chapter 9). The approach of using vegetation maps and inventories or direct mapping of ecotopes, which is in principle superior, was impossible in practice because available data were insufficient. For the same reason, we could not retain all the distinctions with regard to vegetation structure, but instead clustered ecotope types into a smaller number of ecotope groups. Thus, only aquatic, herbaceous and woody vegetations were distinguished for use in DEMNAT. For reasons of simplicity, we shall only use the terms ecotope and ecotope type.
The present (1975-1990) and past (before 1950) distribution and relative species richness of ecotopes were established for 15 selected ecotope types of wet and moist sites from the floristic database FLORBASE (Groen *et al.*, 1992; Groen *et al.*, 1994), using 1187 of the species (Witte & Van der Meijden, 1992). Examples of the present distributions of two ecotope types thus established are given in Plates 9.3 and 9.4 (pages 124 and 125).

In the geographical database, a connection was made between the *presence* of ecotopes of a certain type and the *surface area* of ecoseries. Such a connection is needed to determine the share of the various spatial units in each gridcell for the further calculations.

10.3.2 Dose-effect functions

Dose-effect functions describe the change in the relative number of species belonging to the same ecological species group in relation to a range of hydrological doses. They were defined by Van der Linden *et al.* (1992; see also Runhaar *et al.*, 1996) for all existing combinations of ecoseries type and ecotope type and for all relevant doses.
The dose-effect functions were defined stepwise. Firstly, for each ecoseries type the effects of hydrological doses were estimated in terms of changes to be expected in individual operative site factors. Then, for each ecotope type the changes in relative species richness were estimated as a consequence of these changes in operative site factors. Finally, the dose-effect functions thus found were compared with available empirical data as a partial calibration.

Figure 10.2 shows a selection of dose-effect functions for a lowering of the spring groundwater level (SGL) in one ecotope type for five different ecoseries types. The dose-effect functions express the relationship between relative species richness and SGL.

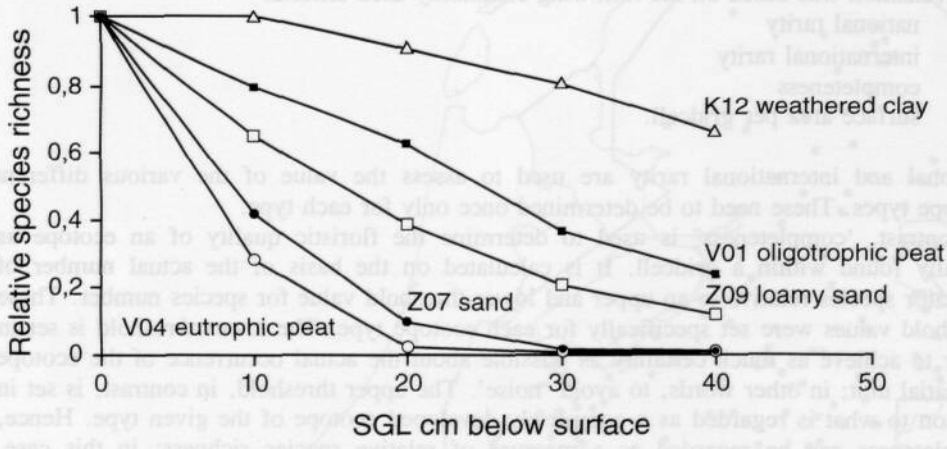

Figure 10.2 Examples of dose-effect functions for lowering of the spring groundwater level (SGL) in terms of a resulting decrease in relative species richness in the ecotope type K21 (herbaceous vegetations of wet, nutrient poor, acid sites) for 5 different ecoseries, here named after the main soil type (Van der Linden *et al.*, 1992).

Despite the fact that dose-effect functions were specified for all combinations of ecotope type and ecoseries type, trials showed biased outcomes because some local circumstances were insufficiently covered by the classifications. Such biased outcomes resulted from, for example, perched water tables, *i.e.* shallow groundwater levels that occur above impermeable layers in the soil. In such cases, groundwater lowering in the region will not affect the local groundwater level, nor the vegetation. Also, subdued response was found to occur when, for example, ditches are so common that plant species can easily shift down the ditch banks when the water level is lowered.

Unexpected process-response conditioning of this kind was taken care of either seperately in DEMNAT, or by improving the dose-effect functions. To this end, the following additional characteristics were specified for each ecoseries type (Klijn *et al.*, 1992):

- probability of occurrence of upward seepage;
- probability of sufficient available soil water in the profile to bridge dry periods;
- probability of occurrence of a perched water table;
- abundance of ditches.

The specification of these characteristics — all divided in few, coarse, classes — enabled improvement of the predictions.

10.3.3 Nature valuation system

A specific procedure was developed for nature valuation (Witte & van der Meijden, 1992). This procedure should not only enable the predicted effects to be expressed in normative terms, but also, and even more importantly, allow the results for each gridcell to be summed in one unifying term.

The valuation was based on the following commonly used criteria:

- national rarity
- international rarity
- completeness
- surface area per gridcell.

National and international rarity are used to assess the value of the various different ecotope types. These need to be determined once only for each type.
In contrast, 'completeness' is used to determine the floristic quality of an ecotope as actually found within a gridcell. It is calculated on the basis of the actual number of indicator species relative to an upper and lower threshold value for species number. These threshold values were set specifically for each ecotope type. The lower threshold is set in order to achieve as much certainty as possible about the actual occurrence of the ecotope as spatial unit; in other words, to avoid 'noise'. The upper threshold, in contrast, is set in relation to what is regarded as a completely developed ecotope of the given type. Hence, completeness can be regarded as a measure of relative species richness; in this case, relative to the thresholds. Completeness was determined from FLORBASE according to a method developed by Witte & Van der Meijden (1992).
The surface area on which an ecotope can be found is also specific to each gridcell. This is based on the abiotic site maps, drawn up as described in Chapter 9. Consequently, both the actual completeness and the surface area of a suitable abiotic site type could be used to calculate nature values for individual gridcells.

The sum of the values established for all the ecotopes in a gridcell weighted according to the value of the respective types to which they belong, provides a measure of the total value.

10.4 An example: scenarios for public and industrial water supply

From 1991 to 1993 an environmental impact assessment was carried out (Beugelink *et al.*, 1992; 1993) for the national Policy Plan on Drinking and Industrial Water Supply (Ministry of VROM, 1993). In this context, the ecological impacts of various scenarios for groundwater extraction in the next decades were analysed. The scenarios are specified in Table 10.1 (page 142), whereas the extraction sites are shown in Figure 10.3.
The majority of the scenarios involve a decrease in groundwater extraction, which reflects the political desire to survey the possibilities for counteracting drought damage to nature values.

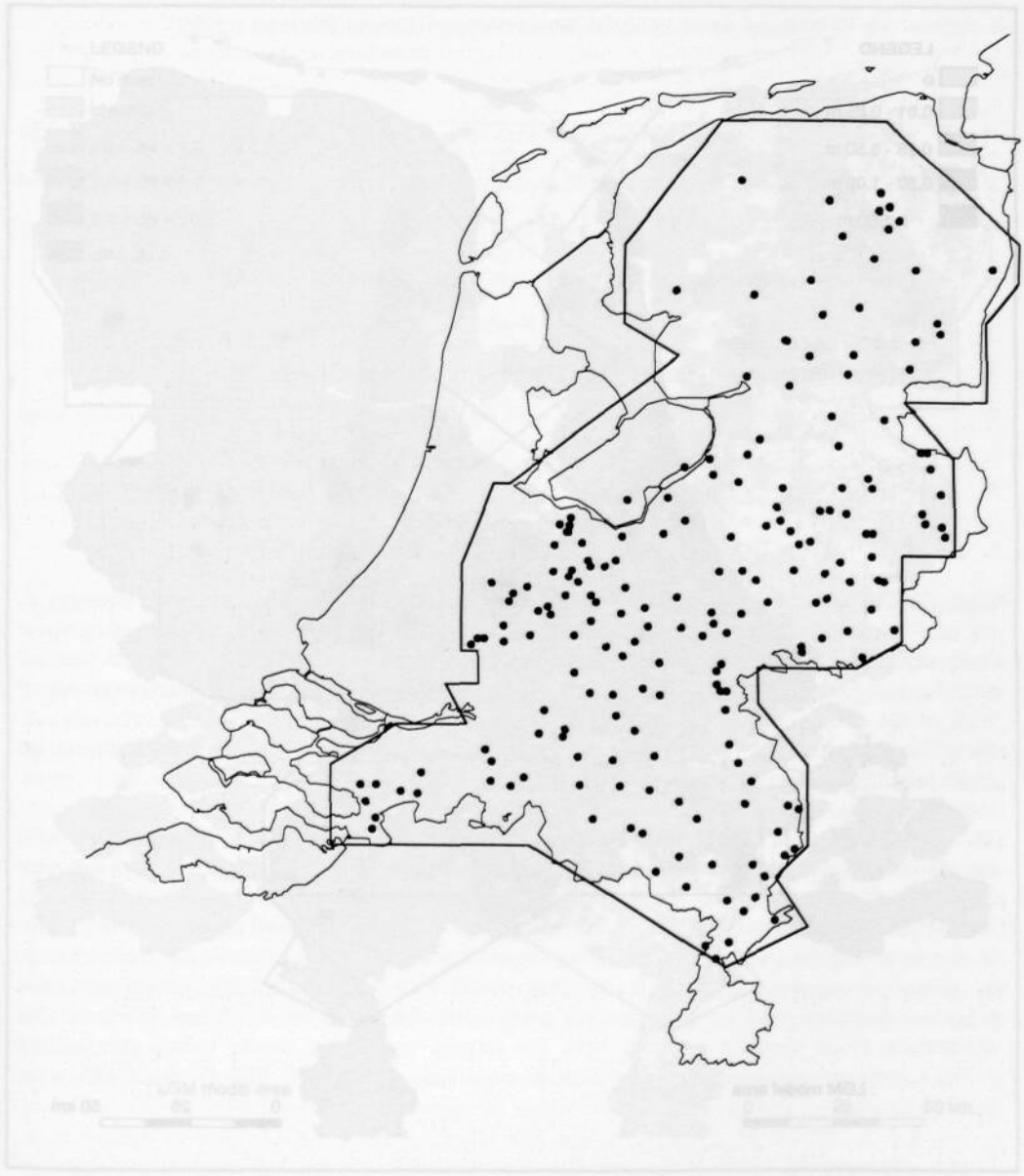

Figure 10.3 Extraction sites for public water supply and industrial water supply in the modelled area, which covers about 75% of the Netherlands.

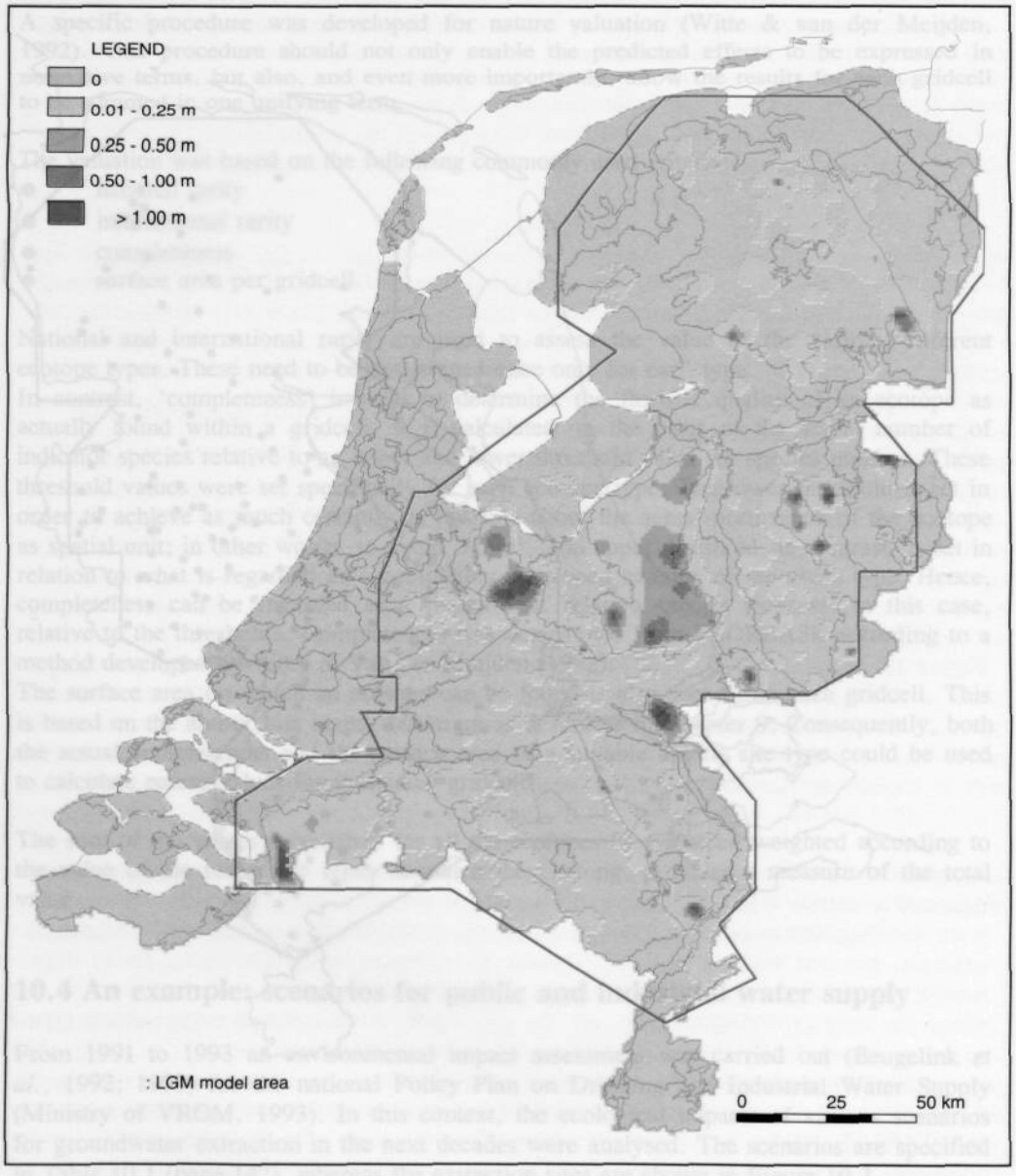

Plate 10.1 The rise of the groundwater level in the modelled area calculated for scenario 3, *i.e.* a 50 % decrease of groundwater extraction for the public water supply (Beugelink *et al.*, 1992).

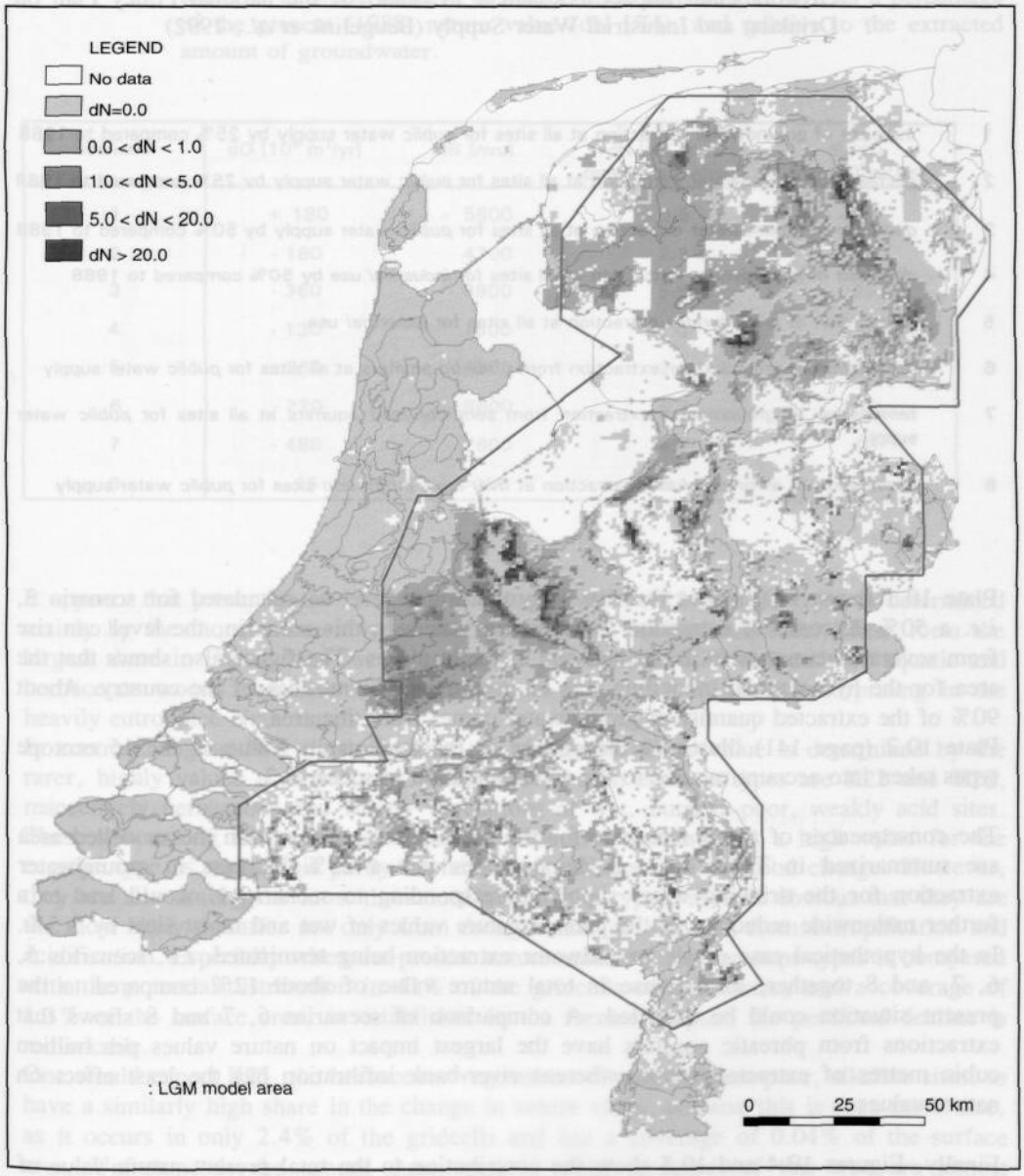

Plate 10.2 Change in the total nature value of ecotopes of the 15 selected types, calculated for scenario 3, relative to the present situation (Beugelink *et al.*, 1992).

Table 10.1 Scenarios for groundwater extraction in the Netherlands as specified for the environmental impact assessment in behalf of the national Policy Plan on Drinking and Industrial Water Supply (Beugelink *et al.*, 1992)

1	*increase* of groundwater extraction at all sites for *public* water supply by 25% compared to 1988
2	*decrease* of groundwater extraction at all sites for *public* water supply by 25% compared to 1988
3	*decrease* of groundwater extraction at all sites for *public* water supply by 50% compared to 1988
4	*decrease* of groundwater extraction at all sites for *industrial* use by 50% compared to 1988
5	*termination* of groundwater extraction at all sites for *industrial* use
6	*termination* of groundwater extraction from *phreatic* aquifers at all sites for *public* water supply
7	*termination* of groundwater extraction from *semi-confined* aquifers at all sites for *public* water supply
8	*termination* of all groundwater extraction at *river-bank infiltration* sites for *public* water supply

Plate 10.1 (page 140) shows the rise of groundwater level as calculated for scenario 3, *i.e.* a 50% decrease of extraction for drinking water. In this scenario, the level can rise from several decimetres to more than one metre in places. The figure also shows that the area for the hydrological modelling was limited to only about 75% of the country. About 90% of the extracted quantity of groundwater comes from this area.
Plate 10.2 (page 141) illustrates the change in the total nature value of the 15 ecotope types taken into account relative to the actual situation in scenario 3.

The consequences of the 8 scenarios for the total nature value within the modelled area are summarized in Table 10.2. It demonstrates that a 25% increase in groundwater extraction for the drinking water supply, corresponding to scenario 1, would lead to a further nationwide reduction of the present nature values of wet and moist sites by 3.5%. In the hypothetical case of all groundwater extraction being terminated, *i.e.* scenarios 5, 6, 7, and 8 together, an increase in total nature value of about 12% compared to the present situation could be expected. A comparison of scenarios 6, 7 and 8 shows that extractions from phreatic aquifers have the largest impact on nature values per million cubic metres of extracted water, whereas river-bank infiltration has the least effect on nature values.

Finally, Figures 10.4 and 10.5 show the contribution to the total present nature value of each ecotope type, respectively the change in nature value in scenario 3 for each ecotope type.

Table 10.2 The predicted consequences of the eight scenarios (first two columns) in terms of absolute changes in nature value units (dN [nvu]), as a percentage of the present (1988) nature value (dN [%]), and relative to the extracted amount of groundwater.

scenario	dQ [10^6 m^3/yr]	dN [nvu]	dN [%] i.r.t. present situation	dN/dQ [nvu/$10^6$$m^3$/yr]
1	+ 180	- 5600	- 3.5	- 31
2	- 180	4700	2.9	26
3	- 360	8900	5.5	25
4	- 130	1500	1.0	11.5
5	- 260	3300	2.0	13
6	- 220	8600	5.3	39
7	- 480	7600	4.7	16
8	- 25	200	0.1	8

It appears that presently the total nature value of all the 15 ecotope types is determined mainly by the more common types, *i.e.* ecotopes of nutrient-rich sites. These form the largest share in Figure 10.4, as demonstrated by A18 and K28, respectively aquatic and herbaceous woody vegetations of wet, nutrient-rich sites. This is not surprising in the heavily eutrophied Netherlands.

In contrast, Figure 10.5 illustrates that the *change* in nature value is determined by the rarer, highly valued ecotope types. Examples of such ecotope types are K22 and H22, respectively herbaceous and woody vegetations of wet, nutrient-poor, weakly acid sites. These are among the rarest site types at present, which implies a high value for the ecotope type as such. This partly explains their influence on the total change. However, they are also among the most threatened abiotic site types, as it is obvious that they are the most likely to suffer not only from drought damage, but also from eutrophication and acidification. Especially when the present nature value of these ecotope types is compared with the potential distribution in 41% of the gridcells of the country and a coverage of 1.3% of the surface area, as established in the former chapter, the predicted decline is remarkable.

One would expect K23, the herbaceous vegetations of wet, nutrient-poor, alkaline sites, to have a similarly high share in the change in nature value, because this is even more rare, as it occurs in only 2.4% of the gridcells and has a coverage of 0.04% of the surface area. Unfortunately the coastal dune area, in which ecotopes of this type are almost exclusively found, was outside the modelled area.

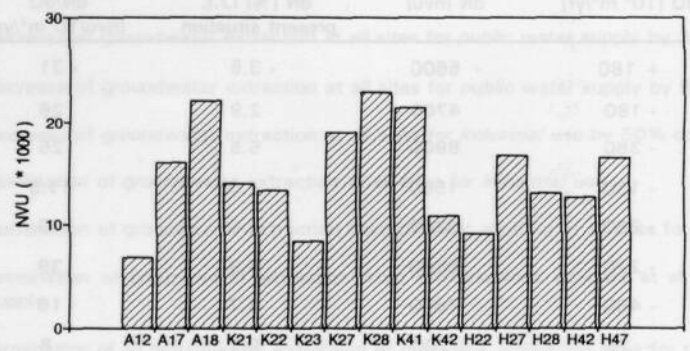

Figure 10.4 Present nature value of ecotopes of each of the 15 selected types for the whole modelled area, derived from floristic species finds in the period 1975-1990. The largest contribution to the total nature value of wet and moist sites comes from the nutrient-rich and moderately nutrient-rich sites (17, 18, 27, 28).

Explanation of codes:

A12	Aquatic vegetation of nutrient-poor weakly acid waters
A17	Aquatic vegetation of moderately nutrient-rich waters
A18	Aquatic vegetation of very nutrient-rich waters
K21	Herbaceous vegetations of wet, nutrient-poor, acid sites
K22	Herbaceous vegetations of wet, nutrient-poor, weakly acid sites
K23	Herbaceous vegetations of wet, nutrient-poor, alkaline sites
K27	Herbaceous vegetations of wet, moderately nutrient-rich sites
K28	Herbaceous vegetations of wet, very nutrient-rich sites
K41	Herbaceous vegetations of moist, nutrient-poor, acid sites
K42	Herbaceous vegetations of moist, nutrient-poor, weakly acid sites
H22	Woody vegetations of wet, nutrient-poor, weakly acid sites
H27	Woody vegetations of wet, moderately nutrient-rich sites
H28	Woody vegetations of wet, very nutrient-rich sites
H42	Woody vegetations of moist, nutrient-poor, weakly acid sites
H47	Woody vegetations of moist, moderately nutrient-rich sites

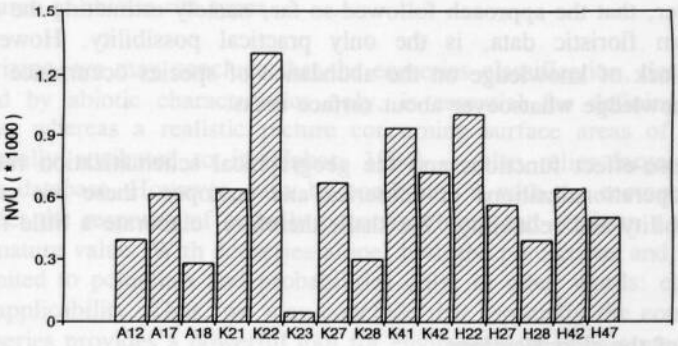

Figure 10.5 Change in the nature value of ecotopes of each of the 15 selected types, calculated for the whole modelled area for scenario 3. The largest changes occur in ecotopes of weakly acid, nutrient-poor sites (22), which depend on upward seepage. Explanation of codes: see figure 10.4

10.5 Discussion

The above example demonstrates that the 2.0 version of DEMNAT is a practical hydro-ecological prediction model for environmental impact assessment at the national level. It yielded useful results with clear differences in the ecological impact of different groundwater extraction scenarios for the next few decades. However, at the request of the responsible ministries only quite simple scenarios have been simulated in the given example. These give some insight into the general relationship between groundwater extraction and nature value at the national scale. Owing to the simplicity of the scenarios, the change in nature value shows a linear relationship with the increase or reduction of each type of groundwater extraction at the national scale (Beugelink *et al.*, 1992, page 60).

However, both the present geographical database with its 1 km^2 grid and the differentiated dose-effect functions allow for a much more detailed impact assessment, enabling applications such as the spatial optimization of groundwater withdrawal from a nature conservation point of view, involving a re-allocation of wells from one region to another, or the assessment of the efficiency of various measures to counteract drought damage (*cf.* Beugelink & Claessen, 1995).

Nevertheless, DEMNAT 2.0 had a number of shortcomings, recognized by the various specialists engaged in its development. The minor shortcomings are being taken care of in version 2.1, but some more important shortcomings remain. These are partly fundamental and partly practical.
They are primarily related to, among other things, a still inadequate validation of the dose-effect functions (Van der Linden *et al.*, 1992; Runhaar *et al.*, 1996) and a lack of detailed geographical data, especially concerning the vegetation and its species composition. It seems, however, that the approach followed so far, namely estimating the occurrence of ecotopes from floristic data, is the only practical possibility. However, this approach implies a lack of knowledge on the abundance of species occurrence (Witte *et al.*, 1992) and no knowledge whatsoever about surface areas.

Because both the dose-effect functions and the geographical schematization rely on the classifications and operationalizations of ecoseries and ecotopes, these are crucial for DEMNAT's applicability and reliability. We shall, therefore, elaborate a little further on their significance.

10.5.1 On the role of the classifications

Firstly, we wish to emphasize that an adequate ecosystem classification is indispensable for any predictive modelling of this kind. As pointed out in Chapter 2, it should allow the response of ecosystems to human interventions to be quantified as well as the significance of this response in terms of changes in nature value.

In the case of DEMNAT, the ecosystem's response could be calculated thanks to the fact that the ecoseries classification permitted quantification of the change in operative site factors, whereas the ecotope classification permitted quantification of the subsequent response of species composition. In this sense, we may conclude that both the ecoseries classification and the ecotope classification served as real, and indispensable, interfaces. As for the significance of the ecoseries classification, the differences between the dose-effect functions for one and the same ecological species group (Figure 10.2) are a good ilustration in this respect.
We may, of course, question whether a simple soil classification would not have served the same purpose. Perhaps it would, but it certainly would not have allowed a proper spatial connection of ecological groups of plant species — of which we only knew the occurrence in gridcells from FLORBASE, but not their precise location — to abiotic spatial units that were properly mapped. The neccessary spatial linkage of species to soil and groundwater in an ecologically meaningful way requires connected classifications of ecotopes and ecoseries. It cannot be achieved by superposition of species or syntaxa (plant communities) onto soil map units, as knowledge on their relationship is insufficient.

For a normative assessment of nature value and changes in nature value, floristic data proved indispensable qualifiers. Thanks to the ecological species groups that are specified for all ecotope types, the floristic database FLORBASE could be used to at least *approximate* floristic nature value in terms of completeness. A further quantification of the nature value of a gridcell by combining the completeness with surface areas of ecotopes proved impossible, because no adequate nationwide vegetation maps are available. Again, we

approximated an assessment on this point by presuming that the *potential* site maps derived from the ecoseries database as described in Chapter 9 are indicative for *actual* surface areas. It is obvious that this is no more than a rough estimate. It is an estimate, however, which owes its confidence to the fairly good correspondence between the occurrence of species from specific ecological species groups on the one hand, and the potential distribution of abiotic operative site types on the other (see Section 9.3.2). Without the ecoseries, we thus would not have had access to any information on the surface area of abiotic site types.

Summarizing, we may conclude that the ecoseries classification, despite the fact that it is classified by abiotic characteristics only, is essential for defining relevant dose-effect functions, whereas a realistic picture concerning surface areas of 'ecotopes' — in the sense usually attributed to it (Haber, 1994) — also relies largely on the nationwide ecoseries database. However, only by combining it with the ecotopes classification could we predict the response of actually occurring biota and perform a quality assessment of current nature value. With ecoseries alone, both the predictions and valuation would have been limited to potentials and probabilities only; in other words: ecoseries alone have a limited applicability. Thus, we may conclude that especially the combination of ecotopes and ecoseries provides a powerful tool for environmental impact prediction, whereas their 'spatial overlay' yields a fairly realistic picture of what are generally considered to be 'ecotopes' (*cf.* Section 3.3).

11 Discussion, conclusions and perspectives

11.1 On the the contribution of this thesis to an ecosystems approach in the environmental sciences

This thesis is about integrated ecosystem classification and mapping for the purpose of environmental management. Thus, it is primarily an attempt to operationalize the ecosystem concept in such a way that it becomes a practical tool for policy analyses. We adopted an integrated approach, because a proper understanding of the environment's functioning as an ecosystem is considered essential for sound environmental management; only by understanding ecosystem functioning can we predict ecosystem response to human activities and trace management opportunities.
In the introduction, we recognized that for a full integration we would have to purposively connect at least three systematic entrances for analyzing the environment as an ecosystem, *viz. processes* (*i.e.* natural and man-induced environmental changes and their time dimensions), *components* (vertical structure) and *patterns* (horizontal structure and scale).

In order to understand the environment's functioning as an ecosystem we must define the relevant components and processes (fluxes of energy, matter and information) as well as all the connections between these. This is of prime importance for any systems approach. Its main aim is to gain insight into the structure and functioning of a system by defining all relevant *linkages*. However, as ecosystems consist of innumerable components with innumerable relationships among them and can be studied at many spatial and temporal scale levels, we soon recognized that we might drown in complexity by striving to develop one model encompassing all linkages within all ecosystems at all spatial scales. However, we were initially saved from really drowning by the 'laws' of hierarchically arranged *organizational levels* (Feibleman, 1954; Rowe, 1961; Schultz, 1967), which prescribe that a level's significance can be understood in relation to the level above, whereas its analysis requires stepping down one level only (*cf.* also O'Neill, 1988; Jørgensen, 1988). For ecosystems this implies that their significance can be understood in relation to the society-environment system above, and their analysis should be based on the recognition of components no lower than, for example, soil or vegetation.
To get ashore, we advocated and consistently followed a deductive approach from theory, based on the recognition of two connected *hierarchies* which are relevant for an integrated ecosystems approach.
Firstly, we postulated a hierarchy of main ecosystem components, which was adopted and slightly adapted from earlier studies (particularly Bakker *et al.*, 1981 elaborating on, *inter alia*, Van der Maarel & Dauvellier, 1978). This hierarchy concerns a *ranking* of ecosystem components according to, among other things, the main direction of fluxes and their consequent decreasing importance as a *controlling factor* for ecosystem development from a process-functional point of view. The primary aim of this hierarchy was to structure the multitude of linkages. Obviously, this ranking ensures the connection between *processes* and *components*.
Secondly, we argued for the use of this component hierarchy as a guideline for classification at different spatial scale levels, because it was recognized that the component ranking

is reflected by patterns of ecosystems of different size, nested within each other. Connected with this, we advocated using *controlling factors as classification characteristics* for a series of related ecosystem classifications at different spatial *scale* levels, primarily because this would yield classifications which would be relevant from a process-functional point of view. The classification guideline reads that the prime controlling factors are likely to be characteristics of the (sets of) component(s) related to a certain spatial scale level/classification level (Figure 4.3). Obviously, this scaling ensures the connection between *patterns* and *components*, and thanks to the hierarchical ranking also between *patterns* and *processes*.
Thus, summarizing, the essence of our contribution is that we used a hierarchical *ranking* of ecosystem components to find the controlling factors which should serve as classification characteristics for a *scaling* of hierarchically nested ecosystems. Or in other words: we contribute a purposive linkage of a process-functional hierarchy (the ranking) to a spatio-temporal hierarchy (the scaling) (*cf.* J.A. Klijn, 1995) in order to achieve a series of functionally related ecosystem classifications for a range of practically relevant spatial scale levels.

11.1.1 On the remainder of this chapter

The deductive approach we have followed is based on hierarchical *concepts* and (conceptual) *models* which structure existing (empirical) knowledge. The approach again results in models: classifications and maps, *i.e.* abstract and iconic models of reality which also structure knowledge.
As for all concepts and models, the question of whether they are 'true' is not relevant, but we may require that they are *theoretically sound* and *consistent with empirical findings*. As these aspects were consistently taken into account in our deductive reasoning, they will not be discussed any further in this chapter. Instead, we shall focus on aspects of the *applicability* of our conceptual models, hierarchies and classification guideline, as well as of the derived classifications and maps in comparison with other approaches. Concordingly, the main question is: do our classifications serve the tasks they were designed for, and do they do so any better than alternative approaches?

As looking back causes things to appear in reverse order, the remainder of this chapter will begin with the *applications* of our classifications (Section 11.2), pass along the *classifications* themselves (Section 11.3), and then towards the guidelines for classification by *controlling factors* and the hierarchies recognized in this context (Section 11.4). In this way, each step of our deductive approach can be evaluated with respect to its consequences for practical applications. Next, we shall formulate the main *conclusions* of our exercise (Section 11.5), after which we shall end with some remarks on the perspectives of an *integrated* ecosystems approach as such (Section 11.6).

11.2 On some practical applications of the classifications and maps

In Chapter 2 we argued that ecological land classification and mapping are first and foremost tools for further analyses. In this thesis, we gave examples of *physical land*

evaluations by using ecodistricts for susceptibility assessment and ecoseries for potential site mapping. Also, the combination of ecoseries and ecotopes was used in a predictive model for *environmental impact assessment*. Here, we shall first reflect on these practical applications.

11.2.1 On applying the ecodistricts for susceptibility assessment

A susceptibility assessment of ecological land units should rely on quantifications or estimates of parameters which control counteracting or triggered processes. The more quantitatively and reliably this can be done, the better it can sustain environmental management.

In the case of assessing the susceptibility of ecodistricts (Chapter 7), the main problem we encountered concerned the fact that the assessment parameters, *i.e.* the parameters that control the relevant processes, are primarily characteristics of other components than those used as classification characteristics. They belong especially to dependent components according to the hierarchical ranking (Figure 4.1), *viz.* (ground)water and soil. This implies that the advantage of using controlling factors for the classification is not self-evident.
However, we could establish the value and variability of the assessment parameters on the basis of correlations between the controlling factors used for the classification and dependent ecosystem characteristics (*cf.* Figure 4.1). These correlations allowed a fair quantification of most assessment parameters, though we needed to rely frequently on component data (*e.g.* Van Duijvenbooden & Breeuwsma, 1987) and disciplinary analyses. Additionally, we could not achieve a really quantitative approach because a number of parameters could only be 'qualified'. This was the case for flooding and upward seepage, for example, both factors with a pronounced influence on many environmental themes.

Meanwhile, various more sophisticated prediction models have yielded an increased accuracy with respect to the quantification of a number of soil chemical processes for a number of environmental themes. Thus, critical loads and/or critical levels have been quantified for, for example, acidification (*cf.* De Vries, 1994). Whether the increased accuracy with respect to part of the system's behaviour also enhances the applicability of the results is questionable, however. As only soil parameters are taken into account in these detailed models, whereas other factors, such as upward seepage and flooding, are neglected, the increased partial accuracy is necessarily attained at the cost of width of scope.
In our opinion, here we come across one of the practical advantages of the use of ecodistricts; our top-down approach ensures that, in principle, all possibly relevant processes and parameters for a certain environmental hazard are taken into account. Moreover, we can trace the points at which further quantification or data gathering is most urgently needed.

Since the ecodistrict susceptibility assessment (Klijn, 1988), much effort has also been put into increasing the spatial detail of nationwide maps concerning, for example, the susceptibility to nitrate leaching or the leaching of atrazine (a pesticide) to the groundwater (RIVM, 1991). The increased spatial accuracy relies on the immediate use of compo-

nent data, especially soil data, which yield the majority of assessment parameters. However, apart from again having a limited scope, the gain in detail should be relativized. Firstly, in general the maps reflect the pattern of the ecodistricts, but more importantly, in our opinion the increased spatial accuracy does not contribute to greater applicability, because the environmental themes concerned require national or regional management.
In this context, perhaps the main advantage of the ecodistricts map becomes manifest, as it ensures a single, relatively multi-purpose, geographical basis for analyzing a range of environmental themes at a spatial scale level relevant for a specific management level (see also Chapters 5 and 7).

11.2.2 On the application of ecoseries for potential site mapping

The ecoseries were used for potential site mapping (Chapter 9), yielding nationwide overviews which showed very satisfactory overlaps with actual site occurrences established from floristic indications. The satisfactory overlaps are most likely due to the purposive design of the ecoseries classification. As the ecoseries classification was designed by starting with the existing ecotope classification as a basis, the classification characteristics of the ecoseries are precisely the factors that control the operative site conditions for the vegetation; one could even state that the ecoseries were elaborated by bottom-up reasoning. Concordingly, the linkage between the controlling factors, *i.e.* the classification characteristics of the ecoseries, and the dependent characteristics, *i.e.* the operative sites and the potential vegetation series depending on these, is very close indeed.

As the 2.0 version of the ecoseries classification and operationalization is still based primarily on the soil map of the Netherlands, one might question whether it has any advantages in comparison to a component classification of soil alone.
Firstly, however, we recall that the soil map of the Netherlands is not really a single component map, but rather a partially integrated land classification which already contains very detailed information on, for example, groundwater levels. Secondly, we recall that we used the LKN-derivative of the soil map and not the soil map itself; this derivative contains even more ecologically relevant additional information on, for example, flooding and the proximity of the sea (De Waal, 1992), which enhances its more-than-one-component character (*cf.* Chapter 8). In converting this into ecoseries, we found many of these added non-soil aspects to be especially relevant. Thirdly, the ecoseries classification is much simpler than both these soil maps, as all ecologically insufficiently relevant information is discarded. Consequently, it is easier to apply for practical management questions as it requires less specialist knowledge. Finally, experiences with the 2.1 update demonstrate that adding upward seepage as a classification characteristic significantly enhances the reliability of the potential site maps (Klijn *et al.*, 1996).
Summarizing, the 2.0 version of the ecoseries classification is already much more integrated than one might imagine at first glance, and the 2.1 version is an improvement on this. Being based on all relevant controlling factors, it has the advantage of being more ecologically relevant than any single-component classification, while also being more simple to understand and apply.

This feature of being simple to apply also holds for the conversion of ecoseries maps (or databases) into maps (or databases) of potential sites. This conversion relies on the so-called site diagrams, which we specified for each ecoseries type (see Chapter 9). These site diagrams have a somewhat *probabilistic* character, in that they define the internal variability of the ecoseries we may expect with respect to the occurrence of operative sites of various type. Firstly, this internal heterogeneity logically follows from our own postulate that ecoseries contain smaller, spatially nested, ecotopes. Secondly, and more importantly, however, it also follows from the experience that operative sites are not very stable in time and gradually evolve with vegetation succession as natural acidification proceeds and the nutrient status increases. This implies that one has to take into account a certain amount of inaccuracy. Consequently, we regard the probabilistic character of the ecoseries an advantage rather than a drawback in comparison with the common deterministic approach to defining potential vegetations, as it takes into account both spatial variability and uncertainties related to vegetation development.

11.2.3 On the application of ecoseries in prediction models, illustrated by DEMNAT

The ecoseries were also applied in the predictive model DEMNAT, in combination with the ecotope classification (Chapter 10). In this case, the ecoseries proved an indispensable interface between man-induced hydrological changes and the response of the vegetation, because the ecoseries classification is not only well-linked to the ecotope classification, but also to the hydrological changes that affect ecosystems. More specifically, the ecoseries cover the characteristics that control the abiotic processes which determine the operative site conditions (*cf.* Figure 10.2). We may therefore conclude that there is a good process-functional linkage with both man-induced environmental changes and the final response variable, *viz.* vegetation.

Just as in the previous section, we may question whether the existing soil classification would not have served the same purpose. In this context, we first refer to the remarks made on the integrated character of the soil map itself in the previous section, but immediately add that the ordinary soil classification certainly would not have allowed a proper spatial connection of ecological groups of plant species on the one hand — of which we only knew the occurrence in gridcells but not their precise location (see Chapter 10) — to abiotic spatial units that were properly mapped on the other. The spatial linkage of species to sites, required by the nature of the nationwide geographical databases, could only be properly accounted for by using connected classifications, in this case of ecotopes and ecoseries. In our opinion it cannot be achieved by a mere superposition of species or syntaxa (plant communities) onto soil map units, as knowledge on their functional linkage is insufficient. This is due primarily to the descriptive character of the commonly used syntaxonomical vegetation classification and its relatively separate elaboration by inductive data processing (see also Section 11.4.1), but it becomes especially manifest in connection with the — for our purpose — unnecessarily complex soil classification, with its relatively strong orientation towards soil genesis.
Consequently, we again hold that ecoseries have a number of practical advantages over component, *i.e.* primarily soil, classifications, also for environmental impact assessment. The same can be remarked about the ecotope classification in comparison with syntaxonomical vegetation classifications.

11.3 On the classifications and maps themselves

In the former sections a number of possibilities and limitations of our classifications and maps were discussed. It appears that the possibilities and problems related to the exemplified classifications are different, which results primarily from the very character of these classifications and maps. The differences are due primarily to differences in the way the ecoregion and ecodistrict classification were established in comparison to how the lower level classifications were developed. We shall briefly reflect on this.

The classifications of ecoregions and ecodistricts were primarily set up as a mapping exercise aimed at 'ecoregionalization' (Chapter 5). Consequently, the classifications lack evident systematics, because they are based largely on geomorphological characterictics and because there was little need for a systematic classification, as the number of map units on nationwide ecoregion and ecodistrict maps are but few. This explains why further analyses based on ecodistricts meet with some problems (see 11.2.1); their main aim was to provide a common, standardized, geographical basis for national environmental management rather than a tool for further analysis (see Chapter 5). This explains why quantitative analyses generally have (also) to rely on data on dependent components with higher spatial accuracy.

In contrast, the classifications of ecoseries and ecotopes are both set up in a much more formal and systematic way, because, firstly, both classifications were primarily developed for effect prediction purposes (see Chapter 8), and, secondly, they were developed as typifications rather than as legends to maps, because an unambiguous diagnostic instrument is required for easy identification of the many thousands to millions of map units for the entire country, either by others or on the basis of existing maps (in our case).

11.3.1 On the application of the guideline for classification by controlling factors

Our series of related classifications for spatially nested ecosystems is based on the presumption that we can use the hierarchy of ecosystem components as a guideline for selecting the controlling factors that would form the most suitable classification characteristics at the different spatial scale levels, while at the same time the classifications would be functionally related by means of exactly this same component hierarchy.
In this thesis we applied this guideline to three classification levels, *viz.* ecoregions, ecodistricts and ecoseries, thus adding to a classification of ecotopes developed *avant la lettre* according to the same guideline by our colleagues (Runhaar *et al.*, 1985; Stevers *et al.*, 1987; Runhaar & Udo de Haes, 1994).

In general, we could adhere to our general classification guideline fairly consistently. However, with the classification of ecoregions and ecodistricts (Chapters 5 and 6) it appeared that the actual classification procedure at these spatial scale levels may be quite 'holistic' and qualitative. This implied a comparatively loose application of the guideline and a relatively weak functional linkage; in fact, the functional linkage merely concerns some genetic linkages between parent material, geomorphology and hydrology within our climatic zone. It seems that this poor functional linkage causes the higher classification

levels to be less systematic *sui generis*, as the relationship between the controlling factors at these scale levels on the one hand, and biota on the other, is always quite indirect.
The classifications of ecoseries and ecotopes, in contrast, were developed in a much more systematic way, which can be explained by the closer connectedness of biota and controlling factors at these spatial scale levels. This allowed a thorough analytical deduction of controlling factors. Consequently, as remarked above (see Sections 11.2.2 and 11.2.3), the process-functional linkage between ecoseries and ecotopes is also quite strong.

Obviously, it is necessary to elaborate an ecosection classification, which should form the linkage between ecodistricts and ecoseries, in order to find out whether indeed 'both ends' can be functionally linked.

11.3.2 On spatial and systematic[1] hierarchies in classification

From the examples in this thesis it becomes clear that none of our classifications gives equal coverage of all the relevant ecosystem components as defined in Figure 4.1. Instead, each classification level is specific for a range of spatial scales only, and is based on characteristics of a limited number of components, whereas characteristics of other components are covered only indirectly by the functional, and subsequently correlative, relationships expressed in the component ranking.
A relevant question in this context is whether it would not be possible — and more worthwhile — to develop *one single* ecosystem classification with a systematic[1] hierarchy covering all components and all spatial scales levels instead. As in soil science (FAO, 1988 or US Soil Taxonomy) or plant sociology (*cf.* Schaminee *et al.*, 1995) such a classification might have different taxonomical levels and would automatically result from a clustering by similarity.
One of the reasons to argue for one single ecosystem classification irrespective of spatial scale is the supposedly better prospects of it being extended towards a world-embracing classification. Whether our approach has any prospects on this point will be discussed in the next section; we shall first explore whether development of our approach towards one single classification with a systematic hierarchy is possible and desirable.

The only justified way to turn our classifications into one single ecosystem classification with balanced emphasis on all components would consist of constituting an overlay (combining the classifications) of ecotopes on ecoseries on ecosections, etc., in order to achieve one single world-embracing classification with a systematic hierachy; *i.e.* of ecotopes!
Firstly, we recall our earlier statement (Section 3.4.1) that in practice a classification with a systematic hierarchy generally cannot easily be applied for multi-scale mapping, because a clustering of small patches with similar features does not yield larger units but only groups of units of the same size as before which are no longer distinguished as separate classes (*cf.* also Zonneveld, 1994, page 45; Zonneveld, 1995, Chapter III). This poses vast problems for small-scale (reconnaissance) mapping, where one generally lapses into

[1]We use the term systematic in this section as defined in Section 3.4.1, *i.e.* in relation to the hierarchies of relatedness as recognized in the *systematics* or *taxonomy* of organisms, vegetations or soils.

the recognition of 'chorological complexes' (from micro-chores to mega-chores) with similar contents in terms of similar mosaics of 'constituent ecotopes'. Earlier, we already expressed the opinion that the recognition of larger spatial units requires a change of view, in that emphasis should be given to other, sometimes concealed, controlling factors, for example climatic differences. By combining all our classifications into one single ecosystem classification with a systematic hierarchy such a change of view would be possible as long as the systematic hierarchy exactly mirrors the spatial hierarchy: each higher classification level — in the systematic sense — would then match a higher scale level.

However, not only would a combination of our classifications produce an unmanageable number of ecotope types, but also the types would be similar in many respects, they would be defined by an enormous number of characteristics — the sum of all the characteristics used for the present range of classifications — and each type would be very limited to almost unique in its occurrence. Although we would get closer to the prevailing conception of an ecotope (Leser, 1991; Haber, 1994), we would not comply with the primary aim of classification, which is bringing order to the apparent chaos of the real world's complexity.

Against these considerations, we regard a series of related classifications focused on mapping hierarchically nested ecosystems at different spatial scale levels more fruitful from a practical point of view than an attempt to develop one single classification with a systematic hierarchy throughout. Each classification can then be used either separately or in combination with one or more classifications of adjacent levels, depending on the practical requirements of a desired application. Obviously, in such an approach the linkage between classifications of adjacent levels needs paramount attention.

11.3.3 On perspectives for an extension towards a series of world-embracing classifications

Obviously, the classifications of the upper spatial scale levels, *i.e.* of ecozones and immediately below, are intended specifically for the global scale (*cf.* Schultz, 1995). However, whether the middle and lower levels also allow extension to other parts of the world is often questioned.

As for the middle levels, *i.e.* ecoregions to ecosections, an extension to other parts of the world appears well possible, as can be deduced from the many examples of similar or at least comparable classifications and mappings. We may refer to, *inter alia*, the ecodistricts map of Belgium (Antrop *et al.*, 1993), the ecoregions of the continental United States (Omernik, 1987; 1995), the various classifications in Canada and the United States (Bailey *et al.*, 1985) or the land system classifications in Australia. These examples partly rely on similar approaches and partly have merely comparable results. This implies that an extension to world-embracing classifications at these levels is theoretically possible, although we recognize that substantial harmonization and due effort is required to make the various approaches really compatible, with special emphasis on enhancing the systematics of the classifications at these levels.

For the lower levels, *i.e.* especially the ecoseries and ecotopes levels, it may seem that our classifications rely on functional relationships between plant species composition and operative site factors, and between operative and conditioning site factors, respectively, which are valid only for the Netherlands. This would inhibit extending the classification to other parts of the world.
In our opinion, however, the principles of classification are certainly of general validity, whereas the classification characteristics may also prove generally valid. The latter opinion is sustained by the increasing convergance of evidence and the resultant widespread consensus about which operative site factors are the most decisive for the species composition in large parts of the world (*cf.* Hills, 1952; Russell, 1973; Ellenberg, 1979), whereas there is also increasing agreement about which soil characteristics are the most decisive conditioning factors for these operative site factors (Jenny, 1941; Kemmers, 1993). Of course, some additional site factors can be expected to be relevant in other parts of the world, whereas the species composition of vegetations in far away areas will naturally be quite different. The latter, however, merely requires defining corresponding ecological species groups with local or regional validity. The possible necessity of adding classification characteristics for ecotopes or ecoseries can be determined only by trying the approach in other parts of the world; this does not fundamentally disrupt the approach as such, however.

11.3.4 On mapping scales

Along with our nomenclature proposal, we specified indicative mapping scales (Chapter 3). In practice, mapping scales are determined by a number of interfering and partly conflicting requirements. In addition, the nature of the tract of land to be mapped may cause practical dilemmas.

We experienced, for example, practical mapping problems when trying to show the coastal dunes as one spatial unit on the ecoregions map, because it forms a rather narrow fringe along the coast. More in general, we estimate that all areas with complex physiography may pose problems in practical mapping. For instance, the internal heterogeneity of the Netherlands' coastal dunes will probably form a problem for mapping in accordance with the indicated mapping scales at all classification levels down from ecosections. In contrast, there are also areas which are so homogeneous that it seems overdone to use large mapping scales for ecotopes. This is the case in, for example, the new polders or, in other parts of the worlds, prairies, savannas, deserts, etc.

We should remember, however, that, as each classification level is related to certain (combinations of) ecosystem components, each level is — while remaining an integrated ecological land classification — conceptually different from the other levels, in that it is based on a specific set of classification characteristics. This conceptual difference between the classification levels relates to the hierarchy of ecosystem components, which ensures the process-functional relationship with adjacent classification levels.
In our opinion, the conceptual consistency of each classification level should have primacy over practical mapping considerations, because the applicability of the classifications from a process-functional perspective should have primacy. Consequently, instead of giving in on conceptual consistency, we rather advocate somewhat easy compliance with

our guideline on mapping scales; the specified mapping scales should indeed be regarded as indicative and certainly not as a straitjacket.

11.4 On the use of controlling factors for ecosystem classification

We have used *controlling factors* as classification characteristics for the various classifications in this thesis. In doing so, we follow Strahler (1975, page 243), who stated: '*A fundamental principle of scientific classification is that the setting up of classes is better done according to the causes of the class differences than according to the effects that differences produce*'. Some are opposed to such an approach, recalling that only characteristics that are internal to the system may be used for classification purposes (*e.g.* Zonneveld, 1994).
However, these controlling *factors*, while certainly being factors in as far as they control processes within the ecosystem, are also *internal* ecosystem *characteristics*, because we defined ecosystems so as to also include abiotic components. Soil texture is a factor for soil moisture, but at the same time an ecosystem characteristic; soil moisture is a factor for plant growth, but at the same time a characteristic; etc. Only properties without ecological relevance are not factors, but then they are also irrelevant for landscape-ecological or environmental research. In fact, the term factor — as we use it — is merely a 'functional' equivalent of the word characteristic. This implies that we cannot discern any conceptual obstacles to using controlling factors as classification characteristics.

Strahler (1975), with his plea for the use of controlling factors as classification characteristics (see above), was referring especially to geomorphology, where landforms are usually classified on the basis of their genesis: eskers, kames, sandrs, sølle, graben, etc. He advocated this approach primarily for *scientific* classification, as here the understanding of causes is of prime importance. Indeed, the scientific taxonomy of life is also based on this approach, stressing the evolutionary ties among species.
However, in contrast to the majority of geomorphologists and bio-taxonomists we are not driven only by scientific motives of better understanding nature, but also, or even rather, by the wish to develop *practical* tools to deal with environmental management questions. This requires knowledge about cause-effect relationships between ecosystem components, as well as about the linkages between human activities and ecosystem components. Consequently, the understanding of cause-effect relations is also of paramount importance from an applications point of view, but, in contrast, we emphasize the current functional relations rather than the historical genetic ties. This explains our choice for a deductive approach and emphasis on current controlling factors.

11.4.1 On an inductive versus a deductive approach

First and foremost we wish to emphasize again that both an *a priori* integrated classification based on deduction, *i.e.* our approach, and an *a posteriori* clustering according to an inductive approach contain arbitrary elements. In contrast to the general feeling, the outcome of an inductive classification — whether by hand or automated procedure — also depends largely on subjective choices, in particular concerning the sample (Bunce *et al.*,

1981; Bunce, 1994; Runhaar *et al.*, 1994): its size, the way it is stratified and the characteristics taken into account are all aspects to be decided upon.

Still, the majority of ecological research and ecological land classification is based on an inductive approach. It is claimed that this yields 'natural' units, which better reflect reality. A disadvantage, however, is that these units are not immediately related to controlling factors. In an inductive approach, one starts without clear hypotheses about controlling factors. These are discovered only afterwards by applying a discriminant analysis to the sample data or by sight analysis of classified or ordinated data. In our opinion, this approach is very valuable for an initial exploration of discriminant or controlling factors — supposing one had no sufficiently clear notion — but one may easily be misled because the size of the sample and the population/area from which it is retrieved may affect the outcome. If, for example, a sample were to be taken from marsh and fen ecosystems in a small region in the north-east of the Netherlands, it would be very likely that nutrient availability and acidity would come out as discriminant factors. When, in contrast, an equal number of samples were to be taken from the western European marshes and fens, one may well find that climate and exposure are the discriminant factors explaining the variation. Against this background, we give preference to basing hypotheses about relationships between ecosystem components on the ecological research carried out so far, *i.e.* on the accumulated knowledge, rather than on a new inductive procedure.
More importantly, a classification which is achieved by inductive data processing remains 'descriptive', as long as the controlling factors are not used as classification characteristics and no hypotheses about their ecological significance are defined. Consequently, such a system has no explanatory or predictive value and has a limited applicability. Of course, correlative relations can be used within the range of the sample, but extrapolation beyond the ranges of one or more factors is by definition unjustified.

In contrast, a classification which is based on deductive reasoning concerning the (strength of) relationships between ecosystem components and characteristics does not have this limitation, because the hypotheses on which it is based are *a priori* of general validity. Therefore, although the degree to which the hypotheses are tested of course determines the validity in practice, we maintain that a deductive approach based on hypotheses concerning ecological relationships is more advantageous from a practical point of view.

11.4.2 On the ranking of ecosystem components

According to many, the analysis of ecological relationships aimed at tracing all the relevant linkages among ecosystem components is preferably done at the beginning of any new case study. However, as we remarked earlier in Chapter 4, this may cause one to get lost in a maze of overwhelming complexity, while it will probably also result in different ecosystem models for each case. In order to obtain a generally applicable, *i.e.* relatively multi-purpose, model, we argued for an ecosystem model reflecting a rule rather than a case (Chapter 4). Such a general model generalizes the most important linkages, in our case by ranking them according to a process-functional hierarchy, which is mirrored by a spatio-temporal hierarchy. The establishment of such a hierarchical ranking may be a

great aid in focusing research efforts on the right components in relation to environmental hazard and spatial and temporal scale.

In our case, we adopted and slightly adapted the component ranking of Bakker *et al.* (1981), empirically established for the Netherlands' coastal dunes by repeated pairwise comparison (*cf.* J.A. Klijn, 1995). We extended it to all the spatial and temporal scale domains relevant for environmental management from the global to the local scale.
It is sometimes questioned whether this ranking is of general validity, because it would differ with local or regional circumstances (*cf.* Van Gils, 1989; Zonneveld, 1989; 1994).

In our opinion, however, there are reasons to regard it as being of sufficiently general validity to justify its use as a structuring concept. Firstly, the many fundamental studies on functional ecological relations can be considered as forming converging evidence as to the soundness of the hierarchy. In this context, we may also regard the substantive worldwide biogeographical research into the relationships between patterns of biota and patterns of controlling factors (see *e.g.* Walter, 1973; Van der Maarel, 1976) as a kind of empirical underpinning. Moreover, there appears to exist adequate consensus among landscape ecologists about the hierachical order, as reflected by its frequent occurrence in many earlier and later works on landscape ecology, if not in figures (Van der Maarel & Dauvellier, 1978; Piket *et al.*, 1987), then in the order of chapters (*e.g.* Vink, 1980).
Secondly, we may regard the results of our evaluation of the ecoregion and ecodistrict maps, as well as of the test of the ecological relevance of the ecoseries, as sustaining the ranking. The test of the ecoseries (Chapter 9) proved that the supposed controlling factors used for the classification, *i.e.* conditioning characteristics of (ground)water and soil, were satisfactory predictors for operative site factors. This sustains the hypothesis that the controlling factors used to classify ecoseries are indeed hierarchically ranked above the controlling factors used to classify ecotopes, or, interpreting matters more liberally, that (ground)water and soil are hierarchically ranked above vegetation. Similarly, the maps of ecoregions and ecodistricts proved to be adequate predictors for a number of, presumedly dependent, abiotic components as stored in the LKN database (see Chapter 6). The evaluation revealed which abiotic components or combinations of components were most closely spatially correlated. As expected, these were geomorphology, soils and groundwater levels, although the match for each component was found to differ with the (stratified) sample. However, we must keep in mind that these evaluations also yield mere converging evidence, as they were not intended to assess the validity of the component ranking as such. But, as Bowler (1992; pages 17-20) explains, converging evidence and scientific consensus is about the most we may expect in ecology and geography.

All together, therefore, we think there is enough circumstantial evidence to be found in the literature to regard the component hierarchy of adequate general validity, whereas also the evaluations of our own classifications add support to the ranking order used.

11.4.3 On the relationship between the component ranking and spatial scales

Even for those who accept the proposed component ranking, it may be questionable whether the relationship with spatial scales is sufficiently strong to use it as a guideline for classification for different scale levels. They retort that all components can be

regarded as hierachically nested systems in themselves, and that their analysis can be carried out at all spatial scales (*cf.* J.A. Klijn, 1995).

From an ecosystems perspective, however, many component characteristics are ecologically insufficiently relevant at the majority of spatial scales; they are either dependent characteristics or just form internal variability. Although we readily admit that each component can indeed be studied at many scale levels, we consider this notion no aid for 'integrated' ecosystem classification and mapping, as it only yields a proper analysis of individual components. Instead, one should decide about factors being more or less important from an ecosystems point of view, even when in fact they all act at the same time and influence one another. This implies that we should discriminate between the controlling altitude and exposure on the one hand, and the responding topoclimate on the other, or between the controlling vegetation structure and the responding microclimate. Moreover, we may again refer to the substantive worldwide biogeographical research into the relationships between patterns of biota and patterns of controlling factors (see the former section), which can also be regarded as empirical underpinning of this relationship (*cf.* also Van der Maarel, 1976).

In other words: we hold the opinion that the acceptance of the process-functional *ranking* of ecosystem components as an aid for structuring research activities requires that one also accepts a relationship between components and spatial and temporal *scaling* as an aid for further scoping the research effort.

11.5 Conclusions

From this thesis and especially the previous discussion we retrieve some general conclusions on the approach we followed and on the classifications we established.

- It was found that the *ecosystem concept* can be operationalized in such a way that it becomes a *practical tool* for analyses in behalf of environmental management. It was also found that this can be done only on the basis of due theoretical considerations and subsequent decisions concerning the research procedure (see below).

- A purposive *connection* of processes, components and patterns, required for a fully integrated operationalization, proved possible by adopting a *systems approach* (*cf.* Von Bertalanffy, 1950; Chorley & Kennedy, 1971), which offers a *common language* detached from individual scientific disciplines. It comprises cause-effect chains which are considered quintessential in the environmental sciences (*cf.* Udo de Haes, 1991), regarding these as process-response systems. A common language for interdisciplinary or transdisciplinary research is of paramount importance for the (physical) environmental sciences.

- The recognition of *organizational levels* (*e.g.* Miller, 1975; Haber, 1982; 1994) — different from system levels —, in combination with the 'Laws of integrated levels' (Feibleman, 1954; *cf.* also Rowe, 1961; O'Neill, 1988; Müller, 1992) was found to be a great help in *delimiting* the scope and depth of the required investi-

gations. It obviates too deep disciplinary investigations as well as excursions irrelevant to the question.

- *Hierarchical concepts* proved to be a fine, and perhaps vital, tool for further focussing attention and for structuring the procedure. In this context, a process-functional hierarchy — a *ranking* — of main ecosystem components, and a spatio-temporal hierarchy — a *scaling* of nested ecosystems — appeared to be the most relevant for a multi-purpose and multi-scale operationalization of the ecosystem concept.

- For the development of ecosystem classifications for different spatial scale levels it proved especially profitable to *link* a scaling of ecological patterns, *i.e.* of nested ecosystems of various size, to a ranking of ecosystem components by their hierarchical relationship through both structure and processes. Thus, we could elaborate (part of) a *series of classifications* for nested ecosystems that are also *functionally* connected.

- The use of *controlling factors* as classification characteristics was found to yield *practically applicable classifications*, because these ensure a good linkage with human activities — the causes of environmental change — on the one hand, and with biotic response variables on the other. The latter linkage automatically follows from the requirement that the controlling factors are directly or indirectly ecologically relevant; this is already partly ensured by the relationship to the component ranking, but needs special attention during the actual development of a classification at a certain level.

- For the *lower*-level classifications, *i.e.* ecotopes and ecoseries, the use of *controlling factors* as classification characteristics proved especially fruitful for applications in *environmental impact assessment*, because of the close and relatively unambiguous relatedness of the vegetation to the controlling factors — operative site factors in the case of ecotopes and conditioning site factors in the case of ecoseries.

- For the *higher*-level classifications, *i.e.* ecoregions and ecodistricts, the advantages of using *controlling factors* as classification characteristics were found to be *less obvious for analysis purposes*, because the abiotic processes that control the response to environmental hazards are often determined by components lower in the hierarchy, especially (ground)water and soil. This implies that ecoregions and ecodistricts must be considered rather as useful *regionalizations* for stratification, aggregation/integration and purposeful extrapolation.

- In our evaluation of the *ecoregion* and *ecodistrict maps* we found that they were mapped in a adequately *consistent* way. However, we may also conclude that the result of our knowledge-based all-encompassing procedure cannot possibly be attained by an *a posteriori* integration of (only some) component data.

- The testing of the *ecoseries classification* revealed *significant correlations* between

predicted site conditions and the site conditions indicated by plant species occurrence. The good correlations can be considered to support the approach to classification and to prove the soundness and usefulness of the ecoseries classification as such.

- When looking back on the classifications established thus far, it appears that the *ecoseries* classification has a thorough systematic structure and proves to be a practicable typification, allowing its (relatively) safe and easy application by others, whereas, in contrast, the *ecoregions and ecodistricts* classifications are much less systematic because they were set up primarily as mapping exercises in an attempt to establish standardized ecological *regionalizations*, and also because they are based on predominantly qualitative geomorphological characteristics. Firstly, this may pose problems when extending the classification to other areas and, secondly, it is still questionable whether the functional connectedness of the classifications at the various levels can be assured; the development of an ecosection classification — intermediate between the ecodistricts and the ecoseries — is required to explore this point.

11.6 On the perspectives of an integrated approach to ecosystems

In the introduction to this thesis we wrote that in the environmental management of the Netherlands an integrated ecosystems approach has been evolving since 1984. However, in our opinion this has not yet resulted in a fully comprehensive ecosystems approach of such conceptual clarity that the communication among researchers and between researchers and policy-makers is optimal. Indeed, we hold that the difficulties encountered in operationalizing an ecosystems approach to the environment for management purposes are due primarily to conceptual and, concurrently, communication problems (*cf.* Haber, 1992a).

These problems are connected partly with the complexity and many-sidedness of the research object; ecosystems can be studied at many spatial scales and many time scales, and consequently feature numerous or even innumerable components and even more functional relationships among these. In addition, though, many approaches may be relevant (Chapter 4) depending on the question at stake, whereas also the disciplinary background of the researchers may play a role: geographers may have another view than functional ecologists, etc. Resultingly, although the amount of relevant research in the environmental sciences is very large indeed, the possibility of linking the various approaches is hampered by the lack of a common conceptual basis.
In our opinion, despite a tradition in theory-making and conceptual thinking among ecologists, there are but few *connecting* theories and/or concepts with (potential) *practical* significance (*cf.* Jørgensen, 1992; Müller et al., 1992). In this context, we do not refer to the many attempts to develop '*Weltanschauungssysteme mit Totalanspruch*', as most of these lack communicative power. Probably, the most practical of the currently used connecting concepts for the Netherlands' environmental management are the environmental themes launched by the ministry of VROM *et al.* (1985; see Chapter 1) and the recognition of a connection between these themes and spatial scales (Klijn, 1988; RIVM;

1988). Both the environmental themes and the spatial scales proved very important aids for structuring information on environmental hazards, as illustrated by the communicative success of 'Concern for Tomorrow' (RIVM, 1988; 1989).

Against this background we hold the opinion that integrating the (physical) environmental sciences requires not only an ecosystems approach, but also the consistent elaboration of relatively simple (*cf.* Haber, 1992), comprehensive ordering and structuring concepts. In this thesis, we have tried to contribute to this by proceeding in a deductive way with the emphasis on concepts and conceptual models. To this end, we have used and adapted concepts and models from various origins, trying to match theoretical considerations — such as on organizational levels and hierarchies — with empirical findings and observation — for example, on the relationships between controlling factors and ecosystem patterns — with, again, practical requirements on ecological land classifications.

In this attempt to operationalize the ecosystem concept, we set as our primary aim to make purposive connections between processes (functional relationships), components (vertical structure) and patterns (horizontal structure and scale). We based this connection on the recognition of two related hierarchies, *viz.* a *ranking* of components and a connected *scaling* of nested ecosystems. We hope to have demonstrated that the recognition of these two hierarchies may contribute substantially to a *practical* operationalization of the ecosystems approach to environmental management; an approach, also, that may enhance the communication among scientists and between scientists and policy-makers. Of course, this requires that the concepts are acceptable from a scientific point of view, because of sufficient theoretical soundness and adequate empirical support, whereas they should also be simple enough to be communicative.

In this context, we would like to finish by stating that, in our opinion, despite their being primarily mental constructs, hierarchies deserve greater credit because of their structuring value, not the least with respect to the classification and mapping of spatially nested ecosystems in behalf of environmental management.

References

Anonymus (Arbeitskreis Standortskartierung), 1980. *Forstliche Standortsaufnahme.* 4th Edition. Landwirtschaftsverlag, Münster-Hiltrup.

Anonymus (Arbeitskreis Standortskartierung), in press. *Forstliche Standortsaufnahme.* 5th (revised) edition. Landwirtschaftsverlag, Münster-Hiltrup.

Antrop, M., H. Gulinck, K. Van Looy, G. De Blust, P. Van Ghelue, I. Melkebeke & E. Kuijken, 1993. *Structuurplan Vlaanderen. Deelfacet open ruimte. Eindrapport.* I.N. Ref. A93-103.

Arnborg, T., 1964. *Det Nordsvenska skogstypschemat (The North-Swedish forest type scheme).* (in Swedish). 7th edition. Svenska skogsvårdsföreningen, Stockholm

Arnold R.W., I. Szabolcs & V.O. Targulian (eds.), 1990. *Global soil change.* IIASA, Laxenburg.

Baaijens, G., 1985. Over grenzen. *De Levende Natuur* 86e jrg/3.

Bailey, R.G., 1976. *Ecoregions of the United States.* USDA Forest Service, Intermountain Region, Ogden, Utah.

Bailey, R.G., 1981. Integrated approaches to classifying land as ecosystems. in: Laban, P. (ed.). *Proceedings of the workshop on land evaluation for forestry.* ILRI publ. 28: 95-109, Wageningen.

Bailey, R.G., 1983. Delineation of ecosystem regions. *Environmental Management* 7: 365-373.

Bailey, R.G., 1985. The factor of scale in ecosystem mapping. *Environmental Management* 9: 271-276.

Bailey, R.G., 1987. Suggested hierarchy of criteria for multi-scale ecosystem mapping. *Landscape and Urban Planning* 14(1987): 313-319.

Bailey, R.G., 1989. Explanatory Supplement to Ecoregions Map of the Continents. *Environmental Conservation* 16/4: 307-310.

Bailey., R.G., 1996. *Ecosystem geography.* Springer Verlag, New York, Inc. 204 pp.

Bailey, R.G., S.C. Zoltai & E.B. Wiken, 1985. Ecological regionalization in Canada and the United States. *Geoforum* 16: 265-275.

Bakker T.W.M., J.A. Klijn & F.J. van Zadelhoff, 1981. *Nederlandse kustduinen. Landschapsecologie.* Pudoc, Wageningen.

Bannink, J.F, H.N. Leijs & I.S. Zonneveld, 1973. *Vegetatie, groeiplaats en boniteit in Nederlandse naaldhoutbossen.* Bodemkundige Studies 9, Stiboka, Wageningen.

Barkman, J.J., 1958. *Phytosociology and ecology of cryptogamic epiphytes.* Assen.

Batjes, N.H. & E.M. Bridges (eds.), 1991. *Mapping of soil and terrain vulnerability to specified chemical compounds in Europe at a scale of 1:5M.* ISRIC, Wageningen.

Beek, K.J., 1978. *Land evaluation for agricultural development.* ILRI publication 23, Wageningen.

Begon, M., J.L. Harper & C.R. Townsend, 1986. *Ecology: individuals, population and communities.*

Blackwell Scientific Publications, Oxford/ London/Edinburgh. 876 pp.

Bekhuis, J., R. Bijlsma, A. van Dijk, F. Hustings, R. Lensink & F. Saris (eds.), 1987. *Atlas van de Nederlandse vogels*. SOVON, Arnhem.

Beugelink, G.P., F.A.M. Claessen & J.H.C. Mülschlegel, 1992. *Effecten op natuur van grondwaterwinning t.b.v. Beleidsplan Drink- en Industriewatervoorziening en MER*. RIVM rapport nr. 714305010, Bilthoven/ RIZA nota nr. 92.059, Lelystad.

Beugelink, G.P., F.A.M. Claessen & J.H.C. Mülschegel, 1993. *Calculating the effects of groundwater abstraction on natural vegetation*. Annual report 1992, National Institute of Public Health and Environmental Protection, Bilthoven.

Blandin, P. & M. Lamotte, 1988. Recherche d'une entité écologique correspondant à l'étude des paysages: la notion d'écocomplexe. *Bull. Ecol.* 19/4: 547-555.

Blasco, F., 1988. The international vegetation map (Toulouse, France). Pages 443-460 in Küchler, A.W. & I.S. Zonneveld (eds.), 1988. *Vegetation mapping*. Kluwer Academic Publishers, Dordrecht.

Boersma, W.T., 1995. *Welke natuurwaarden zijn aanwezig in twee proefgebieden in Noord-Brabant?* RIZA werkdocument 95.153X, Lelystad.

Bolsius E.C.A., J.P. Chardon, C.L.G. Groen, W.B. Harms, F. Klijn, J.L. Mulder, Th. Niessen, C.H.J. van Oijen, Th.M.F. Peterbroers, J.G.M. Schouffoer, E.P. Sterling, B.J. Vreeken, R.W. de Waal & M. van 't Zelfde, 1992. *Op weg naar een landsdekkend databestand. LKN fase 3 Nederland: Interimrapport*. LKN rapport 3, Rijksplanologische Dienst, Den Haag.

Boulding, K.E., 1956. *General systems theory: the skeleton of science*. General Systems, Vol. I.

Bowler, P.J., 1992. (The fontana history of) *The Environmental Sciences*. Fontana Press, London. 634 pp.

Braat, L.C., 1992. *Sustainable multiple use of forest ecosystems: an economic-ecological analysis for forest management in the Netherlands*. Thesis, VU, Amsterdam.

Braat, L.C., A.R. van Amstel, A.C. Garritsen, C.R. van Gool, N. Gremmen, C.L.G. Groen, H.L.M. Rolf, J. Runhaar & J. Wiertz, 1989. *Verdroging van Natuur en Landschap in Nederland*. Ministerie V&W, Lelystad.

Brink, A.B.A., J.A. Mabbutt, R. Webster & P.H.T. Beckett, 1965. *Report of the Working Group on land classification and data storage*. Military Engrg. Exp. Establ. Rep. no. 940. Christchurch, England.

Bunce, 1994. The application of quantitative methods of classification to strategic ecological survey in Britain. Pages 173-182 in F. Klijn (ed.) (1994). *Ecosystem classification for environmental management*. Kluwer Academic Publishers, Dordrecht/ Boston/ London.

Bunce, R.G.H., C.J. Barr & H.A. Whittaker, 1981. *Land classes in Great Britain: preliminary descriptions for users of the method of land classification*. Merlewood Research and Development Paper No. 86.

Canters, K.J., C.P. den Herder, A.A. de Veer, P.W.M. Veelenturf & R.W. de Waal, 1991. Landscape-ecological mapping of the Netherlands. *Landscape Ecology 5/3*: 145-162.

Central Statistics Agency (CBS), 1986. Landbouwgebieden. *Kwartaalbericht Milieustatistieken* 86/3.

Chorley, R.J. & B.A. Kennedy, 1971. *Physical Geography. A Systems Approach*. Prentice-Hall International Inc., London. 370 pp.

Christian, C.S. & G.A. Stewart, 1968. Methodology of integrated surveys. in *Aerial surveys and integrated studies. Proc. Toulouse Conf. 1964*, UNESCO, Paris. pp. 233- 280.

Claessen, F.A.M. & J.P.M. Witte, 1991. National water management strategies for conservation and recovery of terrestrial ecosystems. Pages 526-534 in Ravera, O., 1991. *Terrestrial and Aquatic ecosystems: Pertubation and Recovery*. Ellis Horwood Limited, New York etc.

Claessen, F.A.M., F. Klijn, J.P.M. Witte & J.G. Nienhuis, 1994. Ecosystem classification and hydro-ecological modelling for national water management. Pages 199-222 in Klijn, F. (ed.), 1994. *Ecosystem classification for environmental management*. Kluwer Academic Publishers, Dordrecht/ Boston/ London.

Clarke, S.E., D. White & A.L. Schaedel, 1991. Oregon, USA, ecological regions and subregions for water quality management. *Environmental Management* 15: 847-856.

Conseil de l'Europe, 1987. *Carte de la végétation naturelle des Etats membres des Communautés européennes et du Conseil de l'Europe* (texte A. Noirfalise). Publ. no. EUR 10970.

Dale, V.H., R.H. Gardner & M.G. Turner, 1989. Predicting Across Scales. Comments of the Guest Editors of Landscape Ecology. *Landscape Ecology* 3/3-4: 147-151.

Daubenmire, R., 1952. Forest vegetation of northern Idaho and adjacent Washington and its bearing on concepts of vegetation classification. *Ecol. Monogr. 22*: 301-330.

De Bakker, H. & J. Schelling, 1966. *Systeem van bodemclassificatie voor Nederland, de hogere niveaus*. Pudoc, Wageningen.

De Bakker, H. & J. Schelling (ed. D.J. Brus & C. van Wallenburg), 1989. *Systeem van bodemclassificatie voor Nederland, de hogere niveaus*. 2e, gewijzigde druk. Pudoc, Wageningen.

De Gans, W. & J.F.Th. Schoute, 1973. *Fysisch geografische regio's van Nederland*. Instituut voor Aardwetenschappen, Vrije Universiteit, Amsterdam.

De Groot, W.T., 1992. *Environmental science theory*. Elsevier Science Publishers.

De Jongh, J. & O. Vaessen, 1985. *Globale landschappenkaart van Nederland naar visuele kenmerken van water (schaal 1: 1.000.000)*. SWNBL rapport 6a, Utrecht.

De Lange L. & M.A. de Ruiter (eds.), 1977. *Biologische waterbeoordeling; methoden voor het beoordelen van Nederlands oppervlaktewater op biologische grondslag*. TNO, Delft. 251 pp.

De Vries, F., 1993. *Een fysisch-chemische karakterisering van de eenheden van de bodemkaart van Nederland, schaal 1: 250.000*. SC-DLO rapport 265, Wageningen.

De Vries, W., 1994. *Soil response to acid deposition at different regional scales. Field and laboratory data, critical loads and model predictions*. Thesis, Wageningen. 487 pp.

De Vries, W. & A. Breeuwsma, 1986. Relative importance of natural and anthropogenic proton sources in soils in the Netherlands. *Water, Air and Soil Pollution* 28(1986): 173-184

De Waal, R.W., 1992. *Landschapsecologische kartering van Nederland: Bodem en grondwatertrappen*. LKN rapport nr. 2/ SC-DLO-rapport 132, DLO-Staring Centrum, Wageningen.

De Wit, T., 1976. *Epiphytic lichens and air pollution in the Netherlands*. Rijksinstituut voor Natuurbeheer, Leersum. 228 pp.

Digby, P., N. Galwey & P. Lane, 1989. *Genstat 5. A Second Course.* Clarendon Press, Oxford.

Dokuchaev Soil Institute, 1963. *Soil-geographical zoning of the USSR.* Jerusalem: Israel Prog. Sci. Transl.

Duchaufour, P., 1982. *Pedology: pedogenesis and classification.* (English edition, translated by T.R. Paton), George Allen & Unwin, London. 448 pp.

Duel, H., J.L. Fiselier, F. Klijn & C. Kwakernaak, 1989. *Gebiedsvreemd water in Nederland. Een verkenning van de problematiek van gebiedsvreemd water en de ruimtelijke oplossingsmogelijkheden.* TNO-SCMO rapport R 89/42, Delft/ CML rapport 52, Leiden.

Ellenberg, H., 1973. Versuch einer Klassifikation der Ökosystemen nach funktionalen Gesichtspunkten in H. Ellenberg (ed.). *Ökosystemforschung.* Springer, Berlin.

Engelen, G.B., 1980. *Indeling van Nederland in geohydrologische regio's.*

Engelen, G.B., J.M.J. Gieske & S.O. Los, 1988. *Grondwaterstromingsstelsels in Nederland.* Staatsbosbeheer, Utrecht.

Evelyn, 1661. *Fumifugium or; the Inconvenience of the Aer and Smoake of London Dissipated; together with some Remedies Humbly proposed.* Paper to King Charles II, London.

FAO, 1976. *A framework for land evaluation.* FAO soils bulletin 32, FAO-UNESCO, Rome.

FAO, 1988. *FAO/ Unesco Soil Map of the World, Revised Legend.* World Resources Report 60, FAO, Rome. Reprinted as Technical Paper 20, ISRIC, Wageningen, 1989.

Farjon, J.M.J., 1982. *Landschapsecologische relaties via het oppervlaktewater op nationaal en regionaal niveau.* De Dorschkamp, rapport 316, Wageningen. 183 pp.

Farjon, J.M.J., 1987. Een fysiotooptypologie voor bosontwikkeling in de Randstad. In W.B. Harms (ed.). *Ecologische infrastructuur en bosontwikkeling in de Randstad.* De Dorschkamp, rapport 484, Wageningen.

Feibleman, J.K., 1954. Theory of integrative levels. *British Journal for the Philosophy of Science* 5: 59-66.

Fiselier, J.L., F. Klijn, H. Duel & C. Kwakernaak, 1992. The choice between desiccation of wetlands or the spread of Rhine water over The Netherlands. *Wetlands Ecology and Management* 2(1992)/ 1/2: 85-93

Forman, R.T.T. & M. Godron, 1986. *Landscape Ecology.* Wiley & Sons, New York.

Geraedts, W.H.J.M., 1986. *Voorlopige atlas van de Nederlandse dagvlinders - Rhopalocera.* Vakgroep natuurbeheer, LU, Wageningen.

Gleichman-Verheijen E.C., H.E. van Capelleveen, J.A. Klijn & J.Th.F. Schoute, 1991. *Naar een ecologische classificatie en beoordeling van bodems.* Publikatie RMNO nr. 54, Rijswijk.

Godron, M., 1994. The natural hierarchy of ecological systems. Pages 69-83 in F. Klijn (ed.). *Ecosystem classification for environmental management.* Kluwer Academic Publishers, Dordrecht/ Boston/ London.

Groen, C.L.G., M. Gorree, R. van der Meijden, R. Huele & M. van 't Zelfde, 1992. *FLORBASE; een bestand van de Nederlandse flora, periode 1975-1990.* CML report 91, Leiden.

Groen, C.L.G., R.A.M. Stevers, C.R. van Gool & M.E.A. Broekmeyer, 1993. *Uitwerking ecotopensysteem fase III.* CML report 49, Leiden.

Groen, C.L.G., R. van der Meijden & J. Runhaar, 1994. The use of floristic data to establish the occurrence and quality of ecosystems. Pages 275-290 in Klijn, F. (ed.), 1994. *Ecosystem classification for environmental management.* Kluwer Academic Publishers, Dordrecht/ Boston/ London.

Guinée, J.B., 1995. *Development of a methodology for the environmental life-cycle assessment of products, with a case study on margarines.* Thesis, RU Leiden.

Haase, G., 1973. Zur Ausgliederung von Raumeinheiten der chorischen und regionischen Dimension. *Petermann's Geographische Mitteilungen* 117: 81-90.

Haase, G., 1976. Die Arealstruktur chorischer Naturräume. *Petermann's Geographische Mitteilungen* 120: 130-135

Haase, G., 1989. Medium scale landscape classification in the German Democratic Republic. *Landscape Ecology* 3: 29-41.

Haber, W., 1982. Naturschutzprobleme als Herausforderung an die Forschung. *Natur und Landschaft* 57/1: 3-8.

Haber, W., 1990. Using landscape ecology in planning and management. Pages 217-232 in I.S. Zonneveld & R.T.T. Forman (eds.), 1990. *Changing landscapes: an ecological perspective.* Springer-Verlag, New York.

Haber, W., 1992. Erfahrungen und Erkenntnisse aus 25 Jahren der Lehre und Forschung in Landschaftsökologie: Kann man ökologisch planen? in Duhme, F., R. Lenz and L. Spandau (eds.), 1992. *25 Jahre Lehrstuhl für Landschaftsökologie in Weihenstephan mit Prof.Dr.Dr.h.c. W. Haber.* Festschrift, Weihenstephan.

Haber, W., 1992a. On transfer of scientific information into political action; experiences from the German Federal Council of Environmnetal Advisors. *INTECOL-bulletin* 20(1992): 3-13

Haber, W. 1994. Systems ecological concepts for environmental planning. Pages 49-67 in F. Klijn (ed.), 1994. *Ecosystem classification for environmental management.* Kluwer Academic Publishers, Dordrecht/ Boston/ London.

Harms, W.B. & J. Roos- Klein Lankhorst (eds.), 1994. *Toekomst voor de natuur in de Gelderse Poort: planvorming en evaluatie.* SC-DLO report 298.1, Wageningen.

Harms, W.B., J.P. Knaapen & J. Roos- Klein Lankhorst (eds.), 1991. *Natuurontwikkeling in de Centrale Open Ruimte.* SC-DLO report 138, Wageningen.

Hettelingh, J-P., R.J. Downing & P.A.M. de Smet, 1991. *Mapping Critical Loads for Europe.* CCE technical report no. 1, RIVM, Bilthoven.

Heukels, H. & R. van der Meijden, 1983. *Flora van Nederland.* 20th ed., Wolters-Noordhoff, Groningen.

Heukels, H. & S.J. van Oostroom, 1975. *Flora van Nederland.* 18th ed., Wolters-Noordhoff, Groningen.

Hills, G.A., 1952. *The classification and evaluation of site for forestry.* Res. Rep. 24. Toronto. 41 pp.

Hills, G.A., 1953. The use of site in forest management. *Forestry Chronicle* 29: 128-136

Hof, G.T.A., 1983. Toelichting bij de fysisch-geografische landschappenkaart van Nederland, schaal 1: 600.000. *Geografisch Tijdschrift* 17: 263-268.

Hornung, M., 1993. Defining soil quality for ecosystems and ecosystem functioning. Pages 201-211 in Eijsackers, H.J.P. & T. Hamers (eds.), 1993. *Integrated soil and sediment research: a basis for proper protection*. Kluwer Academic Publishers.

Hughes, R.M. & D.P. Larsen, 1988. Ecoregions: an approach to surface water protection. *Journal of the Water Pollution Control Federation* 60: 486-493.

Hughes, R.M., T.R. Whittier, C.M. Rohm & D.P. Larsen. 1990. A regional framework for establishing recovery criteria. *Environmental Management* 14: 673-683

Hunsaker, C.T., R.L. Graham, G.W. Suter II, R.V. O'Neill, L.W. Barnthouse & R.H. Gardner, 1990. Assessing ecological risk on a regional scale. *Environmental Management* 14: 325-332.

Isachenko, A.G., 1973. *Principles of landscape science and physical geographic regionalization*. J.S.Massey (ed.), Melbourne, Australia.

Jenny, H., 1941. *Factors of soil formation*. MacGraw-Hill, New York/ London.

Jørgensen, S.E., 1992. *Integration of Ecosystem Theories: A Pattern*. Kluwer Academic Publishers, Dordrecht/ Boston/ London.

Kalkhoven, J.T.R., A.H.P Stumpel & S.E. Stumpel-Rienks, 1976. *Landelijke Milieukartering*. Staatsuitgeverij, Den Haag.

Karrer, G. & W. Kilian, 1990. *Standorte und Waldgesellschaften im Leithagebirge*. Mitteilungen der forstlichen Bundesversuchsanstalt Wien, Heft 165, Vienna.

Kemmers, R.H., 1993. Staalkaarten voor een ecologische landevaluatie. *Landschap 10(1993)/1*: 5-22

Keuning, H.J., 1965. *Het Nederlandse volk in zijn woongebied*. Leopolds, Den Haag. 23 pp.

Kilian, W., 1981. Site classification systems used in forestry. Pages 134-151 in: Laban, P. (ed.), 1981. *Proceedings of the workshop on land evaluation for forestry*. ILRI publication 28, Wageningen.

Klijn, F., 1988. *Milieubeheergebieden. Deel A: Indeling van Nederland in ecoregio's en ecodistricten. Deel B: Gevoeligheid van de ecodistricten voor verzuring, vermesting, verontreiniging en verdroging*. CML report 37, Leiden/ RIVM report no. 758702001, Bilthoven. 183 pp.

Klijn, F., 1988a. *Ecoseries. Aanzet tot een standplaatstypologie*. CML-report 45, Leiden/ DBW-RIZA working document nr. 8.084 X, Lelystad.

Klijn, F., 1989. *Landschapsecologische Kartering Nederland: Grondwaterrelaties*. CML report 51, Leiden/ Stiboka report no. 2107, Wageningen.

Klijn, F., 1991. Hierarchical classification of ecosystems: a tool for susceptibility analysis and quality evaluation for environmental policy. Pages 80-89 in Ravera, O., 1991. *Terrestrial and aquatic ecosystems. Perturbation and recovery*. Ellis Horwood, Chichester.

Klijn, F., 1991a. Environmental Susceptibility to Chemicals: from Processes to Patterns, with Special Reference to mapping Characteristics and Spatial Scales. Pages 9-22 in Batjes, N.H. and Bridges, E.M., 1991. *Proceedings of the International Workshop on Mapping of Soil and Terrain Vulnerability to Specified Chemical Compounds in Europe at a Scale of 1: 5 M*. ISRIC, Wageningen.

Klijn, F., 1994. Spatially nested ecosystems, guidelines for classification from a hierarchical perspective. Pages 85-116 in F. Klijn (ed.), 1994. *Ecosystem classification for environmental management*. Kluwer

Academic Publishers, Dordrecht/ Boston/ London.

Klijn, F. & P.K. Koster, 1988. *Milieubeheergebieden ten behoeve van nationaal gebiedsgericht milieubeleid.* RIVM rapport no. 758702002, Bilthoven.

Klijn, F. & A. Laansma, 1990. *Gebiedsgericht milieubeleid; theorie en praktijk en aanzet tot onderzoeksprogrammering.* CML report 61, Leiden. 75 pp.

Klijn, F. & R.W. de Waal, 1992. Ecologische bodemclassificatie. Een pragmatische aanpak vanuit de standplaatsbenadering. *Landschap 9(1992)/3*: 175-187

Klijn, F. & H.A. Udo de Haes, 1990. Hiërarchische ecosysteemclassificatie; voorstel voor een eenduidig begrippenkader. *Landschap* 1990 7/4: 215-233.

Klijn, F. & H.A. Udo de Haes, 1994. A hierarchical approach to ecosystems and its implications for ecological land classification. *Landscape Ecology* 9: 89-104

Klijn, F. & M. van 't Zelfde, 1992. *Technical document on the digitized maps of ecoregions and ecodsitricts in ARC-INFO.* CML notitie 10, Leiden. 16 pp.

Klijn, F., J.B. Latour, M.I. Nip, C.L.G. Groen, H.A. Udo de Haes, M.M.H.E. van den Berg & J.J. Hofstra, 1990. *De milieukwaliteit van ecodistricten. Deel 2: Methode en aanzet tot uitwerking.* RIVM-rapport 751901003, Bilthoven/ CML mededelingen 63, Leiden, 110 pp.

Klijn, F., A. ter Harmsel & C.L.G. Groen, 1992. *Ecoseries 2.0. Naar een ecoserieclassificatie ten behoeve van het hydro-ecologisch voorspellingsmodel DEMNAT-2.* CML report 85, Leiden/Bilthoven.

Klijn, F., J. Runhaar & M. van 't Zelfde, 1996. *Ecoseries 2.1; verbetering en operationalisatie van een classificatie van ecoseries voor DEMNAT 2.1.* CML report, Leiden.

Klijn, J.A., 1995. *Hierarchical concepts in landscape ecology and its underlying disciplines.* SC-DLO report 100, Wageningen.

Kloosterhuis, J.L. & J.C. Pape, 1976. *Toelichting bij de bodemkaart van Nederland 1: 200.000 ten behoeve van de Landelijke Milieukartering.* Stiboka, Wageningen. 85 pp.

Klopatek, J.M., J.T. Kitchings, R.J. Olson, D.K. Kumar & L.K. Mann, 1981. A hierarchical system for evaluating regional ecological resources. *Biological Conservation* 20: 271-290.

Krzanowski, W.J., 1988. *Principles of multivariate analysis. A users perspective.* Clarendon Press, Oxford.

Küchler, A.W. & I.S. Zonneveld (eds.), 1988. *Vegetation mapping.* Kluwer Academic Publishers, Dordrecht.

Kuylenstierna & M. Chadwick, 1991. The Sensitivity of Soil and Ecosystems to Acidic Depositions. In: Batjes, N.H. & E.M. Bridges (eds.), 1991. *Proceedings of the International Workshop on Mapping of Soil and Terrain Vulnerability to Specified Chemical Compounds in Europe at a Scale of 1: 5 M.* ISRIC, Wageningen.

Kwak, R.G.M., L. Reijrink, P. Opdam & W. Vos, 1987. *Broedvogeldistricten van Nederland: een ruimtelijke visie op de Nederlandse avifauna.* RIN/Dorschkamp-rapport, Wageningen/ PUDOC (1988), Wageningen.

Kwakernaak, C., 1982. *Landscape Ecology of a Prealpine Area. A contribution to the development of a unifying concept in landscape ecology, based on investigations in the La Berra-Schwarzsee area (Fribourg,*

Switzerland). Publ. Fys. Geogr. Bodemk. Lab., UvA, nr. 33. 165 pp.

Lands Directorate Environment Canada, 1981. *Ecological Land Survey Guidelines for Environmental Impact Analysis.* Ecological Land Classification Series no. 13. 42 pp.

Latour J.B. & C.L.G. Groen, 1994. Ecological assessment of natural resources in agroecosystems of Dutch Lowland Peat Area. *Environmental Management* 18: 455-464

Latour, J.B., C.L.G. Groen & M. van 't Zelfde, 1991. *De milieukwaliteit van de ecodistricten het Laagveengebied en de Kalkrijke Duinen.* RIVM-report 711901001, Bilthoven/ CML-report 73, Leiden.

Lenz, R.J.M., 1994. Ecosystem classification by budgets of material; the example of forest ecosystems classified as proton budget types. Pages 117-137 in Klijn, F. (ed.), 1994. *Ecosystem classification for environmental management.* Kluwer Academic Publishers, Dordrecht.

Lenz, R.J.M., 1994a. Consequences of a 150 years' history of human disturbances in some forest ecosystems. Pages 265-289 in Boyle, T.J.B. & C.E.B. Boyle (eds.), 1994. *Biodiversity, Temperate Ecosystems, and Global Change.* Springer Verlag, Berlin/Heidelberg.

Leser, H., 1976. *Landschaftsökologie.* Verlag Eugen Ulmer, Stuttgart.

Leser, H., 1991. *Landschaftsökologie.* 3rd (revised) Ed. Verlag Eugen Ulmer, Stuttgart. 647 pp.

Lovelock, J.E., 1979. *Gaia: a new look at life on Earth.* Oxford University Press, Oxford.

Maas, G.J., R.W. de Waal & H.P. Wolfert, 1994. *Landschapsecologische kartering van Nederland: Geomorfologie.* SC-DLO report 335, Wageningen.

Mardia, K.V., J.T. Kent & J.M. Bibby, 1979. *Multivariate analysis.* Academic Press, London/ Orlando.

Miller, J.G., 1975. The nature of living systems. *Behavioral Science* 20 (1975): 343-365.

Miller, J.G., 1978. *Living Systems.* McGraw- Hill Inc. 1102 pp.

Ministry of VROM, 1983. *Voorlopig Indicatief Meerjarenprogramma Bodem 1984-1988.*

Ministry of VROM, 1984. *Indicatief Meerjaren Programma Lucht 1985-1989.*

Ministry of VROM, 1984a. *Meer dan de som der delen.*

Ministry of VROM, 1990. *Gebiedsgericht milieubeleid, actieplan.* SDU uitgeverij, Den Haag. 49 pp.

Ministry of VROM, 1993. *Beleidsplan Drink- en Industriewatervoorziening. Deel 1: ontwerp planologische kernbeslissing.* Ministerie VROM, Den Haag.

Ministries of VROM, L&V & V&W, 1984. *Indicatief Meerjaren Programma Milieubeheer 1985-1989.*

Ministries of VROM, L&V and V&W, 1985. *Indicatief Meerjarenprogramma Milieubeheer 1986-1990.*

Ministries of VROM, EZ, L&V and V&W, 1989. *Kiezen of verliezen. Nationaal Milieubeleidsplan.*

Ministry of V&W, 1989. *Water voor nu en later. Derde nota waterhuishouding.* SDU uitgeverij, Den Haag.

Ministries of V&W and VROM, 1985. *Indicatief Meerjaren Programma Water 1985-1989.*

Müller, S., 1970. Öko-Serien der baden-württembergischen forstlichen Standortskartierung am Beispiel der Kalkverwitterungslehme. *Mitteilungen der Deutschen Bodenkundlichen Gesellschaft* 10: 43-46.

Müller, F., 1992. Hierarchical approaches to ecosystem theory. *Ecological Modelling* 63(1992): 215-242.

Müller, F., W. Windhorst & S.E. Jørgensen, 1992. Recent problems in ecosystem theory - conclusions of the workshop. *Ecological Modelling* 63(1992): 325-331

National Forestry Service (SBB)/ NMF (Ministry of Agriculture and Fisheries), 1984. *Natuurgebieden, bossen en natuurwaarden in het agrarisch landschap.* Inspectie Natuurbehoud, Den Haag.

Naveh, Z. & A.S. Lieberman, 1994. *Landscape ecology. Theory and application.* 2nd edition, Springer Verlag, New York/ Berlin/ Heidelberg.

Neef, E., 1967. *Die theoretische Grundlagen der Landschaftslehre.* Verlag H. Haack, Gotha-Leipzig.

Neef, E., 1970. Zu einigen Begriffen in der Ökologie. *Archiv für Naturschutz und Landschafsforschung* 10/4: 233-240.

Nienhuis J.G., J.B.S. Gan & R. Lieste, 1992. *Het ecohydrologisch voorspellingsmodel DEMNAT-2, technische modelbeschrijving.* Special series "Onderzoek Effecten Grondwaterwinning" report no. 2, Bilthoven.

Nilsson, J. (ed.), 1986. *Critical loads for nitrogen and sulphur.* Nordisk Ministerråd, Miljørapport 1986:11.

Nip, M.I., J.B. Latour, F. Klijn, P.K. Koster, C.L.G. Groen, H.A. Udo de Haes & H.A.M. de Kruijf, 1992. Environmental Quality Assessment of Ecodistricts: a Comprehensive Method for Environmental Policy. Pages 865-881 in McKenzie, D.H., D.E. Hyatt & V.J. McDonald (eds.). *Ecological Indicators.* Volume 2. Elsevier Applied Science, London & New York.

O'Neill, R.V, 1988. Hierarchy Theory and Global Change. Pages 29-45 in Rosswall, T, R.G. Woodmansee & P.G. Risser (eds.). *Scales and Global Change.* Wiley & Sons Ltd, Chichester/ London.

O'Neill, R.V., D.L. DeAngelis, J.B. Waide & T.F.H. Allen, 1986. *A Hierarchical Concept of Ecosystems.* Princeton University Press, Princeton.

Odum, E.P., 1971. *Fundamentals of ecology.* W.B. Saunders Comp., Philadelphia. 574 pp.

Odum, H.T., 1983. *Systems Ecology, an introduction.* John Wiley & Sons, New York.

Omernik, J.M., 1987. Ecoregions of the conterminous United States. *Annals of the Association of American Geographers* 77: 118-125.

Omernik, J.M., 1995. Ecoregions: a Spatial Framework for Environmental Management. Pages 49-62 in Davis, W.S. & T. Simon (eds.). *Biological Assessment and Criteria: Tools for Water Resource Planning and Decision Making.* Lewis Publishers, Boca Raton/ Ann Harbor/ London/ Tokyo.

Pakes, U., R.H. van Waveren & F.A.M. Claessen, 1992. *Berekening invloed systeemvreemd water met DEMGEN.* RIZA werkdocument 92-117X, Lelystad/ special series "Onderzoek Effecten Grondwaterwinning" report no. 8, Bilthoven.

Pastoors, M.J.H., 1992. *Landelijk Grondwater Model (LGM); conceptuele modelbeschrijving.* RIVM report no. 714305004/ special series "Onderzoek Effecten Grondwaterwinning" report no. 10, Bilthoven.

Pastoors, M.J.H., 1992a. *Landelijk Grondwater Model (LGM); berekeningsresultaten.* RIVM report no.

714305005/ special series "Onderzoek Effecten Grondwaterwinning" report no. 12, Bilthoven.

Pedroli, G.B.M., 1987. Ecohydrologie, een overzicht. *Landschap* 4(1987)/4: 320-330.

Pfister, R.D., 1975. *Land capability assessment by habitat types*. Soc. of Americ. forest., Washington.

Piket J.J.C., J.T.R. Kalkhoven, A.A. de Veer & W. Vos, 1987. *Atlas van Nederland, deel 16: Landschap*. Staatsuitgeverij, Den Haag. 23 pp.

Poore, M.E.D., 1962. The method of successive approximation in descriptive ecology. *Advances in ecological research* 1(1962): 35-68.

Projectgroep Verdroging, 1989. *Verdroging van natuur en landschap in Nederland, beschrijving en analyse*. Ministerie van Verkeer & Waterstaat, Den Haag.

Pulles, J.W., 1985. *Beleidsanalyse van de waterhuishouding van Nederland: PAWN*. Rijkswaterstaat, Den Haag.

Rey, P. & H. Gaussen, 1955. *Service de la carte de la végétation de la France au 200 000*. CNRS, Service de la carte phytogéographique pp 11-34. Toulouse.

RIVM (ed. F. Langeweg), 1988. *Zorgen voor Morgen. Nationale Milieuverkenning 1985-2010*. Samsom H.D. Tjeenk Willink, Alphen aan de Rijn. 456 pp.

RIVM (ed. F. Langeweg), 1989. *Concern for tomorrow. National environmental survey 1985 - 2010*. National Institute of Public Health and Environmental Protection, Bilthoven. 344 pp.

RIVM, 1991. *Nationale milieuverkenning 1990-2010*. RIVM, Bilthoven. 550 pp.

Roelofs, H.J., T.J. Beukeboom, A. Ebregt & W. Vos, 1982. *Landschapecologische relaties via het grondwater op nationaal en regionaal niveau*. De Dorschkamp rapport no. 317, Wageningen. 122 pp.

Rolf, H.L.M., 1989. *Verlaging van de grondwaterstanden in Nederland*. (Ministerie van Verkeer en Waterstaat, 1989. Verdroging van Natuur en Landschap in Nederland. Technisch rapport: Verlaging van de grondwaterstanden in Nederland, analyse periode 1950-1986)

Rowe, J.S., 1961. The level-of-integration concept and ecology. *Ecology* 42(1961)/2: 420-427.

Rowe, J.S. & J.W. Sheard, 1981. Ecological land classification: a survey approach. *Environmental Management* 5: 451-464.

Runhaar, J., 1989. Toetsing ecotopensysteem: relatie tussen de vochtindicatie van de vegetatie en grondwaterstanden. *Landschap* 6(1989): 129-146.

Runhaar, J. & H.A. Udo de Haes, 1994. The use of site factors as ecosystem classification characteristics. Pages 139-172 in Klijn, F. (ed.), 1994. *Ecosystem classification for environmental management*. Kluwer Academic Publishers, Dordrecht/ Boston/ London.

Runhaar, J., C.L.G. Groen, R. van der Meijden & R.A.M. Stevers, 1987. Een nieuwe indeling in ecologische groepen binnen de Nederlandse flora. *Gorteria* 13(1987): 277 - 359.

Runhaar, J., J.P.M. Witte & R.H.G. Jongman. 1994. Ellenberg-indicatiewaarden: verbetering met reciprocal averaging? *Landschap* 11/1: 41-47

Runhaar., J., C.R. van Gool & C.L.G. Groen, 1996. Impact of hydrological changes on nature conservati-

on areas in the Netherlands. *Biological Conservation* (in press)

Schaminée, J.H.J., A.H.F. Stortelder & V. Westhoff, 1995. *De vegetatie van Nederland.* Deel 1. Inleiding tot de plantensociologie - grondslagen, methoden en toepassingen. Opulus Press, Uppsala/ Leiden.

Schultz, A.M., 1967. The ecosystem as a conceptual tool in the management of natural resources. In Ciriacy-Wantrup, S.V. & J.J. Parsons, 1967. *Natural Resources: Quality and Quantity.* University of California Press, Berkeley and Los Angeles.

Schultz, J., 1995. *The ecozones of the world. The ecological divisions of the geosphere.* Springer-Verlag New York, Inc. 460 pp.

Siegel, S. & N.J. Castellan, 1988. *Nonparametric statistics for the behavioral sciences.* Second edition. McGraw-Hill, Inc., New York.

Sinnige, C.A.M., W.L.M. Tamis & F. Klijn, 1991. *Aanzet tot een ecotopenclassificatie toegespitst op de bodemfauna.* CML-report 75, Leiden.

Sinnige, C.A.M., W.L.M. Tamis & F. Klijn, 1992. *Indeling van bodemfauna in ecologische soortengroepen.* CML report 80, Leiden

Sinnige, C.A.M., W.L.M. Tamis & F. Klijn, 1994. *Eigenschappen van ecosystemen voor ecotoxicologische effectvoorspelling.* CML report 104, Leiden.

Smuts, J.C., 1936. *Holism and evolution.* 3rd Ed. McMillan and Co., London.

Soil Survey Staff, 1975. *US Soil Taxonomy.* US Department of Agriculture, Washington DC.

Steur, G.G.L. & W. Heijink, 1987. *Bodemkaart van Nederland, schaal 1: 50.000: Algemene begrippen en indelingen.* 3rd edition, Stiboka, Wageningen.

Steur, G.C.L., F. de Vries, & C. van Wallenburg, 1985. *Bodemkaart van Nederland 1: 250.000.* Stiboka, Wageningen.

Stevers, R.A.M., J. Runhaar, H.A. Udo de Haes & C.L.G. Groen, 1987. Het CML-ecotopensysteem, een landelijke ecosysteemtypologie toegespitst op de vegetatie. *Landschap* 4/2: 135-150.

Stevers, R.A.M., J. Runhaar & C.L.G. Groen, 1987. *Het CML-ecotopensysteem. Uitwerking voor Noord-, West- en Zuidwest-Nederland.* CML report 34, Leiden.

Stigliani, W.M. (ed.), 1991. *Chemical Time Bombs: Definition, Concepts, and Examples.* IIASA Executive report 16, Laxenburg, Austria.

Strahler, A.N., 1975. *Physical geography.* Fourth edition 1975. John Wiley & Sons, Inc. 643 pp.

Suter, G.W. II, 1990. Endpoints for Regional Ecological Risk Assessment. *Environmental Management* 14/1: 9-23.

Sverdrup, H., W. de Vries & A. Henriksen, 1990. *Mapping Critical Loads; a guidance to criteria, calculations, data collection and mapping of critical loads.* Miljørapport 1990:14, Nordic Council of Ministers.

Tansley, 1935. The use and abuse of vegetational concepts and terms. *Ecology* 16(1935): 284-307.

Tansley, 1939. *The British isles and their vegetation.* Cambridge University Press.

Teixeira, R.M. (ed.), 1979. *Atlas van de Nederlandse broedvogels.* Natuurmonumenten/ SOVON. 431 pp.

Theurillat, J.P., 1992. L'analyse du paysage végétal en symphytocoenologie: ses niveaux et leurs domaines spatiaux. *Bull. Ecol.* 23/1-2: 83-92.

Tomaselli, R., 1973. *La vegetazione forestale d'Italia.* Collana Verde 33, Roma.

Toth, J., 1963. A theoretical analysis of groundwater flow in small drainage basins. *Journal of Geophysical Research* 68/16: 4795-4812.

Traas, T., F. Klijn & T. Aldenberg, 1995. *Ecosysteemgerichte risicobeoordeling van stoffen. Naar een model ter bepaling van het ecologisch rendement van brongericht stoffenbeleid.* RIVM rapport nr. 719102038, Bilthoven/ CML report 107, Leiden.

Trewartha, G.T., 1968. *An Introduction to Climate.* 4th ed. McGraw Hill, New York.

Troll, C., 1950. *Die geographische Landschaft und ihre Erforschung.* Studium Generale 3, no 4/5: 163-181.

Troll, C., 1968. Landschaftsökologie. In Tüxen, R. (ed.). *Pflanzensoziologie und Landschaftsökologie.* Junk, the Hague.

Troll, C., 1970. Landschaftsökologie (geoecology) und Biocoenologie. Eine terminologische Studie. *Rev. Roum. Géol., Géophys. et Géogr.: Série de Géographie* 14/1: 9-18.

Tüxen, R., 1978. Versuch zur Sigma-Syntaxonomie mitteleuropäischer Flusstal-Gesellschaften. Pages 273-286 in *Ber. Int. Symp. Int. Vereinigung Vegetationsk.*

Tüxen, R., 1978a. Bemerkungen zu historischen, begrifflichen und methodischen Grundlagen der Synsoziologie. Pages 3-11 in *Ber. Int. Symp. Int. Vereinigung Vegetationsk.*

Turner, M.G., V.H. Dale & R.H. Gardner, 1989. Predicting across scales: Theory development and testing. *Landscape Ecology* 3/3-4: 245-252.

Udo de Haes, H.A., 1991. Milieukunde, begripsbepaling en afbakening. Pages 21-34 in Boersema, J.J., J.W. Copius Peereboom & W.T. de Groot (eds.), 1991. *Basisboek milieukunde.* 4th ed. Boom, Meppel.

Udo de Haes, H.A. & F. Klijn, 1994. Environmental policy and ecosystem classification. Pages 1-21 in Klijn, F. (ed.), 1994. *Ecosystem classification for environmental management.* Kluwer Academic Publishers, Dordrecht/ Boston/ London.

Ulrich, B., 1980. Production and consumption of hydrogen ions in the ecosphere. Pages 255-282 in T.C. Hutchinson & M. Havas, 1980. *Effects of acid precipitation on terrestrial ecosystems.* Plenium Press, New York/ London.

UNEP/ISRIC, 1990. *World map on status of human-induced soil degradation.* UNEP/ISRIC, Nairobi, Kenya.

Urban, D.L., R.V. O'Neill, H.H. Shugart, 1987. Landscape ecology: a hierarchical perspective can help scientists understand spatial patterns. *Bioscience* 37(1987)/2: 119- 127.

USDA (Soil Survey Staff), 1977. *US Soil Taxonomy, 7th approximation.* US Department of Agriculture, Washington DC.

Van Beusekom C.F., J.M.J. Farjon, F. Boekema, B. Lammers, J.G. de Molenaar & W.P.C. Zeeman, 1990. *Handboek grondwaterbeheer voor Natuur, Bos en Landschap.* SDU, Den Haag.

Van der Linden, M.G.A.M., J. Runhaar & M. van 't Zelfde, 1992. *Effecten van ingrepen in de waterhuishouding op vegetaties van natte en vochtige standplaatsen.* CML report no. 86, Leiden/ Special series "Onderzoek Effecten Grondwaterwinning" report no. 7, Bilthoven.

Van der Linden, M.G.A.M., C.L.G. Groen & M. van 't Zelfde, 1995. *Landschapsecologische kartering Nederland: vegetatie en landschapselementen.* LKN report 10, Den Haag/CML report 108, Leiden.

Van der Maarel, E. & P.L. Dauvellier, 1978. *Naar een Globaal Ecologisch Model voor de ruimtelijke ontwikkeling van Nederland.* Staatsuitgeverij, The Hague.

Van der Maarel, E., 1976. On the establishment of plant community boundaries. *Ber. Deutsch. Bot. Ges.* 89 (1976): 415-443.

Van der Voet, E., 1996. *Substances from cradle to grave. Development of a methodology for the analysis of substance flows through the economy and the environment of a region, with case studies on cadmium and nitrogen compounds.* Thesis, RU Leiden.

Van de Ven, G.P., et al., 1985. *Atlas van Nederland, deel 15: Water.* Stichting Wetenschappelijke Atlas van Nederland, Staatsuitgeverij, Den Haag.

Van de Weg, R.F. (ed.), 1981. *Veldwerksyllabus voor de richtingen 3 en 4.* Fysisch Geografisch en Bodemkundig Laboratorium, Universiteit van Amsterdam. 103 pp.

Van Duijvenbooden, W. & A. Breeuwsma (eds.), 1987. *Kwetsbaarheid van het grondwater.* RIVM report 840387003, 69 pp.

Van Gils, H.M., 1989. Legends of landscape ecology maps. *ITC Journal* 1989/1: 41-48.

Van Staalduinen, C.J., 1979. *Geologische overzichtskaart van Nederland 1: 600.000.* Rijks Geologische Dienst, Haarlem.

Van Wirdum, G., 1979. *Dynamic aspects of trophic gradients in a mire complex.* Pages 66-82 in Proceedings and Informations no. 25, TNO- Committee on Hydrological Research, The Hague.

Van Wirdum, G., 1984. *Water related impacts on nature protection sites.* Pages 27-57 in Proceedings and Information no. 34, TNO- Committee on Hydrological Research, The Hague.

Várallyay, G. & M. Leszták, 1990. Susceptibility of soils to physical degradation in Hungary. *Soil Technology* 3(1990): 289-298

Várallyay, G., M. Redly & A. Muranyi, 1989. Map of the susceptibility of soils to acidification in Hungary. Pages 79-94 in I. Szabolcs (ed.), 1989. *Ecological impact of acidification. Proceedings of the Symposium ' Environmental Threats to Forest and other Natural Ecosystems".* Hungarian Academy of Sciences (MTA), Budapest.

Veelenturf, P.W.M. (ed.), 1987. *Landschapsecologische kartering van Nederland, Fase 1.* Studierapport 39, Rijksplanologische Dienst. Staatsuitgeverij, Den Haag.

Veen, A.W.L., 1982. Specifying the concept of landscape cell (ecotope) in terms of interacting physico-chemical processes and external vegetation characteristics. *Proc. Int. Congr. Neth. Soc. Landscape Ecol. Veldhoven, 1981.* Pudoc, Wageningen, 1992.

Verdonschot, P.F.M., J. Runhaar, W.F. van der Hoek, C.F.M. de Bok & B.P.M. Specken, 1992. *Aanzet tot een ecologische indeling van oppervlaktewateren in Nederland.* RIN-report 92/1/ CML-report 78, Leersum.

Verstraten, J.M., 1982. *De bodem als bufferend systeem tegen verzuring.* Inaugurele rede Universiteit van Amsterdam. 16 pp.

Vink, A.P.A., 1963. *Aspects de pédologie appliquée.* La Bacconnière, Neuchâtel.

Vink, A.P.A., 1975. *Land Use in Advancing Agriculture.* Springer Verlag, Berlin/ Heidelberg/ New York. 394 pp.

Vink, A.P.A., 1980. *Landschapsecologie en landgebruik.* Bohn, Scheltema & Holkema, Utrecht. 151 pp.

Vink, A.P.A., 1983. *Landscape ecology and land use.* Edited by D.A. Davidson. Longman, London and New York.

Visscher, H.A., 1972. *Het Nederlandse landschap. Een typologie ten behoeve van het milieubeheer.* Het Spectrum, Utrecht. 155 pp.

Visscher, H.A., 1975. *De Nederlandse landschappen. Ontstaan, wetenschappelijke betekenis, belevingswaarde.* Aula Paperback 32, Het Spectrum, Utrecht/Antwerpen.

Von Bertalanffy, L., 1950. An Outline of General System Theory. *The British Journal for the Philosophy of Science* 1 (1950-1951): 134-165.

Vos, C.C. & P. Opdam (eds.), 1993. *Landscape ecology of a stressed environment.* Chapman & Hall, London.

Vos, W., W.B. Harms & A.H.F. Stortelder (eds.), 1982. *Vooronderzoek naar landschapsecologische relaties tussen ecosystemen.* De Dorschkamp rapport no. 246, Wageningen. 209 pp.

Wagner, A., 1968. Ökoserie, Ökoseriengruppe und Standortsgruppe, neue Begriffe in der forstlichen Standortsaufnahme. *Allgemeine Forstzeitschrift 42*: 731-732

Walter, H., 1973. *Vegetation of the earth in relation to climate and the eco-physiological conditions.* Springer Verlag, New York Inc.

Walter, H., 1979. *Vegetation of the earth and ecological systems of the geo-biosphere.* 2nd ed., Springer Verlag, New York Inc.

Westhoff, V. & A.J. den Held, 1969. *Plantengemeenschappen in Nederland.* Thieme, Zutphen.

Wiken, E.B., 1979. Rationale and methods of ecological land surveys: an overview of Canadian approaches. in Taylor, D.G. (ed.), 1979. *Land/ wildlife integration.* Ecological Land Classification Series no. 11, Lands Directorate Environment Canada.

Wiken, E.B. & G.R. Ironside, 1977. The development of ecological (biophysical) land classification in Canada. *Landscape Planning* 4: 273-275.

Wiken, E.B., D.M. Welch, G.R. Ironside & D.G. Taylor, 1981. *The Northern Yukon: An ecological Land Survey.* Ecological Land Classification Series no. 6. 197 pp.

Witte, J.P.M., 1990. *DEMNAT: aanzet tot een landelijk ecohydrologisch voorspellingsmodel.* DBW/RIZA nota 90.057, Lelystad.

Witte, J.P.M. & R. van der Meijden, 1990. *Natte en vochtige ecosystemen in Nederland.* KNNV-mededeling 200, Utrecht.

Witte, J.P.M. & R. van der Meijden, 1992. *Verspreiding en natuurwaarden van ecotoopgroepen in Nederland*. RIVM, Bilthoven.

Witte, J.P.M., C.L.G. Groen & J.G. Nienhuis, 1992. *Het ecohydrologisch voorspellingsmodel DEMNAT-2, conceptuele modelbeschrijving*. CML report 89, Leiden/ RIVM report 714305007, Bilthoven.

Witte, J.P.M., F. Klijn, F.A.M. Claessen, C.L.G. Groen & R. van der Meijden, 1992a. A Model to Predict and Assess the Impacts of Hydrological Changes on Terrestrial Ecosystems, and its use on a Climate Scenario. *Wetland Ecology and Management* 2: 69-83.

Witte, J.P.M., C.L.G. Groen, R. van der Meijden, R. & J.G. Nienhuis, 1993. DEMNAT: a national model for the effects of water management on the vegetation. Pages 31-51 in Hooghart, J.C & C.W.S. Posthumus, 1993. *The use of hydro-ecological models in the Netherlands*. Proceedings and Information/ TNO Committee on Hydrological Research: no. 47.

Zagwijn W.H. & C.J. van Staalduinen (eds.), 1975. *Toelichting bij de geologische overzichtskaarten van Nederland*. Rijksgeologische Dienst, Haarlem.

Zagwijn, W.H. et al., 1986. *Atlas van Nederland, deel 13: Geologie*. Stichting Wetenschappelijke Atlas van Nederland, Staatsuitgeverij, Den Haag.

Zonneveld, I.S., 1979. *Land Evaluation and Land(scape) Science*. ITC Textbook on Photo-Interpretation, Volume VII, 2nd edition, ITC, Enschede.

Zonneveld, I.S., 1984. Landschapsbeeld en landschapsecologie. *Landschap* 1984/1: 5-9.

Zonneveld, I.S., 1989. The land unit. A fundamental concept in landscape ecology, and its applications. *Landscape Ecology* 3(1989)/2: 67-86.

Zonneveld, I.S., 1994. Basic principles of classification. Pages 23-47 in Klijn, F. (ed.), 1994. *Ecosystem classification for environmental management*. Kluwer Academic Publishers, Dordrecht/ Boston/ London.

Zonneveld, I.S., 1995. *Land ecology*. SPB Academic Publishing, Amsterdam.

Zonneveld, J.I.S., 1985. *Levend land; de geografie van het Nederlandse landschap*. Bohn, Scheltema & Holkema, Utrecht/ Antwerpen.

Curriculum vitae

Frans Klijn was born on 16 January 1958 in Gorinchem. He obtained the Gymnasium-β diploma in 1976 at the Gymnasium Camphusianum in Gorinchem. In the same year he began his studies in physical geography at the University of Amsterdam. For his master's degree he specialized in landscape ecology within the main field of physical geography applied to planning (prof. dr. ir. A.P.A. Vink), with soil erosion as second specialization (prof. dr. P.D. Jungerius). Secondary topics comprised applied landscape ecology (prof. ir. P. Tideman), environmental studies and cultural anthropology, all at the University of Amsterdam, as well as plant ecology/ vegetation science at the University of Uppsala, Sweden (prof. dr. E. van der Maarel). Field experience was gained in Switzerland, Austria, Italy, the fen communities of the far northeast of the Netherlands, and Lapland. Additionally, a first degree teaching certificate was obtained. In his last year of studies, he worked as a teaching assistent at the Interfaculty Department of Environmental Science. He graduated *cum laude* in 1984.

In 1984 and 1985 he worked for some months at the Department of Environmental Science of the University of Amsterdam to investigate trash. Subsequently, he was a research and teaching assistent for two years at the Laboratory of Physical Geography and Soil Science of the University of Amsterdam. From 1987 to 1996 he worked at the Centre of Environmental Science of Leiden University (CML), firstly as a researcher and subsequently as a project co-ordinator and university teacher. During this period he was especially engaged in environmental mapping and assessment and geographical environment analyses. He also tutored groups of postgraduates in water management and nature conservation projects in the context of the European Postgraduate Course in Environmental Management (EPCEM). In 1989 he was awarded a prize for his innovative and unorthodox ideas about managing the flood risks along the branches of the Rhine by the then minister of Housing, Physical Planning and Environmental Management (VROM), Nijpels. Since September 1996 he works at Delft Hydraulics, where he is primarily engaged in river basin management.

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List of publications

Fiselier, J.L., F. Klijn et al., 1981. *Landschap en Vaarrecreatie. Een toegepast landschapsecologisch onderzoek naar de mogelijkheden en beperkingen voor vaarrecreatie in Oost-Groningen.* Rapport Fys. Geogr. & Bodemk. Lab. UvA, nr. 10, Amsterdam. 46 pp., 4 losse kaarten.

Klijn, F., 1982. Degradatie van het Mediterrane bos; De maquis; Groeiwijze van niet-houtige gewassen. Pages 962-967 in: *Spectrum Encyclopedie van de Natuur van Europa*, deel IV (1982).

Goderie, R., P. Jans, F. Klijn & R. Leemans, 1985. Dominance diversity curves from Central and Northern European forests. In: *Studies in Plant Ecology*, Vol. 16, Uppsala.

Klijn, F., 1985. *Hergebruik van grof huishoudelijk afval, in het bijzonder in de provincie Noord-Holland. Verkennend onderzoek in opdracht van de Provinciale Waterstaat van Noord-Holland.* IVAM Onderzoeksreeks nr. 15, Amsterdam. 65 pp.

Akkerman, S.S., R. Cuperus, F. Klijn & H.A. Udo de Haes, 1987. *Palen en perken in het verkeersbeleid. Een verkennend onderzoek naar de mogelijkheden om normen en criteria voor milieukwaliteit in te brengen in de planning op het gebied van verkeer en infrastructuur.* CML-mededeling 33, Leiden. 108 pp.

Klijn, F., 1987. *Landschappelijke diversiteit: de relatie tussen biotische en abiotische diversiteit in het Strijper Aa-gebied.* Rapport nr. 32 Fysisch Geografisch & Bodemkundig Laboratorium, Universiteit van Amsterdam. 56 pp.

Klijn, F., 1988. *Milieubeheergebieden.* Deel A: Indeling van Nederland in ecoregio's en ecodistricten; Deel B: Gevoeligheid van de ecodistricten voor verzuring, vermesting, verontreiniging en verdroging. RIVM-rapport nr. 758702001, Bilthoven/ CML-mededelingen 37, Leiden. 183 pp.

Klijn, F., 1988. *Ecoseries: aanzet tot een standplaatstypologie.* CML-mededeling 45, Leiden/ DBW/RIZA-werkdocument nr. 88.084 x, Lelystad. 42 pp.

Klijn, F. & P.K. Koster, 1988. *Milieubeheergebieden ten behoeve van nationaal gebiedsgericht milieubeleid.* RIVM-rapport nr. 758702002, Bilthoven. 21 pp.

Klijn, F. & T.N. Ligthart, 1988. The applicability of remote sensing in habitat network planning on a regional scale; a case study in the province of Groningen, The Netherlands. Pages 179-180 in: K.-F. Schreiber (Hrsg.). *Connectivity in Landscape Ecology.* Münstersche Geographische Arbeiten 29, Münster.

Duel, H., J.L. Fiselier, F. Klijn & C. Kwakernaak, 1989. *Gebiedsvreemd water in Nederland; een verkenning van de problematiek van gebiedsvreemd water en de ruimtelijke oplossingsmogelijkheden.* SCMO-TNO rapport R 89/42, Delft/ CML-mededelingen 52, Leiden. 209 pp.

Klijn, F., 1989. *Landschapsecologische Kartering Nederland: grondwaterrelaties.* CML-mededelingen 51, Leiden/ Stiboka-rapport 2107, Wageningen. 41 pp.

Klijn, F., 1989. Hierarchical classification of ecosystems as a tool for susceptibility analysis and quality evaluation for environmental policy. *S.IT.E Notizie: Bolletino della Società Italiana di Ecologia*, Vol. X, 3-4 (1989): 116.

Klijn, F. & P.T.J.C. van Rooy, 1989. Waterbeheer: sectoraal of integraal? Een concrete vraag in Oost-Twente. H_2O 22(1989)/11: 342-346.

Van den Berg, M.M.H.E. & F. Klijn, 1989. On the quality evaluation of ecosystems for national environmental policy. *S.IT.E Notizie: Bolletino della Società Italiana di Ecologia*, Vol. X, 3-4 (1989): 257

Fiselier, J.L. & F. Klijn, 1990. Withering fields or Rhine water distribution; on the environmental effects of increased Rhine water distribution and alternative stategies. 14 pages in: *Proceedings of the European Conference on Landscape Ecological Effects of Climatic Change*, Lunteren, 1989.

Glasbergen, P. & F. Klijn, 1990. Overview paper: integrated water and wetland management. Pages 683-693 in: M. Marchand & H.A. Udo de Haes (eds.). *The Peoples Role in Wetland Management*; Proceedings of the International Conference on Wetlands, 5-8 June 1989, Leiden.

Klijn, F. & A. Laansma, 1990. *Gebiedsgericht milieubeleid; theorie en praktijk en aanzet tot onderzoeksprogrammering.* CML-rapport 61, Leiden. 75 pp., 109 pp. bijlagen.

Klijn, F., J.B. Latour, M.I. Nip, C.L.G. Groen & H.A. Udo de Haes, 1990. *De milieukwaliteit van ecodistricten. Deel 1: Ecologische normstelling en milieukwaliteitsbepaling.* RIVM-rapport 751901002, Bilthoven/ CML-rapport 62, Leiden. 28 pp.

Klijn, F., J.B. Latour, M.I. Nip, C.L.G. Groen, H.A. Udo de Haes, M.M.H.E. van den Berg & J.J.

Hofstra, 1990. *De milieukwaliteit van ecodistricten. Deel 2: Methode en aanzet tot uitwerking.* RIVM-rapport 751901003, Bilthoven/ CML-rapport 63, Leiden. 110 pp.

Klijn, F. & H.A. Udo de Haes, 1990. Hiërarchische ecosysteemclassificatie: voorstel voor een eenduidig begrippenkader. *Landschap* 7(1990)/4: 215-233

Udo de Haes, H.A. & F. Klijn, 1990. Gebiedsgericht milieubeleid. Pages 49-61 in: Leroy, P. (red.), 1990. *Milieubeleid in de jaren negentig; integratie van grijs en groen?* Congresboek IVMNL-KUN, SSN, Nijmegen.

Udo de Haes, H.A., E. van der Voet, M.I. Nip, F. Klijn, C.L.G. Groen & J.B. Latour, 1990. De opzet van ecologische normstelling voor terrestrische gebieden. Pages 87-108 in: *Strategieën voor ecologische normstelling. Het spel en de knikkers* (red. A.A.A. van der Schraaf, H.A.M. de Kruijf, R. Cuperus), SDU-uitgeverij, 's-Gravenhage.

Appelman, K., W. Alberts, R. Cuperus, C.L.G. Groen, F. Klijn & K.J. Canters, 1991. *Landschapsecologische kartering voor milieu-effectrapportage bij aanleg van rijkswegen. Eindrapport methode-ontwikkeling.* Rijkswaterstaat DWW/ Rijkswaterstaat MD, Delft. 54 pp.

Claessen, F.A.M., J.Ph.M. Witte, F. Klijn, C.L.G. Groen & R. van der Meijden, 1991. Terrestrische Natuur en de Waterhuishouding van Nederland. *H_2O* 24(1991)/12: 330-337

Glasbergen, P. & F. Klijn, 1991. Integrated water and wetland management: towards a project approach. *Landscape and Urban Planning* 20(1991): 257-262

Klijn, F., 1991. Hierarchical classification of ecosystems: a tool for susceptibility analysis and quality evaluation for environmental policy. Pages 80-89 in: O. Ravera (ed.), 1991. *Terrestrial and aquatic ecosystems: perturbation and recovery.* Ellis Horwood Ltd.

Klijn, F., 1991. Environmental Susceptibility to Chemicals: from Processes to Patterns, with Special Reference to Mapping Characteristics and Spatial Scales. Pages 9-22 in: Batjes, N.H. & E.M. Bridges (eds.). *Mapping of Soil and Terrain Vulnerability to Specified Chemical Compounds in Europe at a Scale of 1:5 M: proceedings of an International Workshop held at Wageningen, the Netherlands (20-23 March 1991).* ISRIC, Wageningen.

Klijn, F., C.L.G. Groen & W.L.M. Tamis, 1991. Ecotopenclassificatie ten behoeve van het stoffenbeleid. Pages 267-285 in: G.P. Hekstra & F.J.M. van Linden (red.). *Flora en fauna chemisch onder druk*; verslag van een nationaal symposium georganiseerd door de Nederlandse Ecologenvereniging - Oecologische Kring, Arnhem, 9-10 oktober 1990. Pudoc, Wageningen.

Kwakernaak, C., F. Klijn, J.L. Fiselier & H. Duel, 1991. Alternatieven voor wateraanvoer; een aanzet tot gebiedsgericht beleid tegen de toenemende invloed van gebiedsvreemd water. *Landschap* 8(1991)/2: 93-107

Sinnige, C.A.M., W.L.M. Tamis & F. Klijn, 1991. *Aanzet tot een ecotopenclassificatie toegespitst op de bodemfauna.* CML report 75, Leiden. 60 pp.

Udo de Haes, H.A., M.I. Nip & F. Klijn, 1991. Towards Sustainability: Indicators of Environmental Quality. Pages 89-105 in: Onno Kuik & Harmen Verbruggen (eds.), 1991. *In Search of Indicators of Sustainable Development.* Kluwer Academic Publishers, Dordrecht/Boston/London.

WLO- Werkgroep Integraal Waterbeheer (B.G.H.J. Beltman, J.M.H. Demon, H. Duel, J.L. Fiselier, J. de Jongh, J.F. Jonkhof, F. Klijn, C. Kwakernaak, A.F.M. Meuleman, J. Runhaar, C.C. Vos, M. Weisz & J.A.W. de Wit), 1991. *Water in balans.* Reeks Landschapsstudies 15, Pudoc, Wageningen. 120 pp.

Bolsius E.C.A., J.P. Chardon, C.L.G. Groen, W.B. Harms, F. Klijn, J.L. Mulder, Th. Niessen, C.H.J.

van Oijen, Th.M.F. Peterbroers, J.G.M. Schouffoer, E.P. Sterling, B.J. Vreeken, R.W. de Waal & M. van 't Zelfde, 1992. *Op weg naar een landsdekkend databestand. LKN fase 3 Nederland: Interimrapport.* LKN rapport 3, Rijksplanologische Dienst, Den Haag.

Fiselier, J.L., F. Klijn, H. Duel & C. Kwakernaak, 1992. The choice between desiccation of wetlands or the spread of Rhine water over The Netherlands. *Wetlands Ecology and Management* 2(1992)/ 1/2: 85-93

Groen, C.L.G. & F. Klijn, 1992. The Ecotope System, an Ecosystem Typology focusing on Vegetation. Page 1532 in: McKenzie, D.H., D.E. Hyatt & V.J. McDonald (eds.). *Proceedings of the International Symposium on Ecological Indicators*, held Oct. 16-18, 1990 in Fort Lauderdale, Florida. Elsevier Applied Science, London & New York.

Havinga, R., F. Klijn, V. Loeffen & W. van der Slikke, 1992. Overlaten en groene rivieren: geschiedenis of toekomst? *H_2O* 25(1992)/10: 248-254

Klijn, F. & R.W. de Waal, 1992. Ecologische Bodemclassificatie: een pragmatische benadering vanuit de standplaatsidee. *Landschap* 9(1992)/3: 175-187

Klijn, F., A. ter Harmsel & C.L.G. Groen, 1992. *Ecoseries 2.0. Naar een ecoserieclassificatie ten behoeve van het hydro-ecologisch voorspellingsmodel DEMNAT-2.* CML report 85, Leiden/ druk RIVM, Bilthoven. 77 pp. met 9 bijlagen.

Klijn, F. & M. van 't Zelfde, 1992. *Technical document on the digitized maps of ecoregions and ecodistricts in ARC-INFO.* CML paper 10, Leiden. 16 pp.

Nip, M.I., J.B. Latour, F. Klijn, P.K. Koster, C.L.G. Groen, H.A. Udo de Haes & H.A.M. de Kruijf, 1992. Environmental Quality Assessment of Ecodistricts: a Comprehensive Method for Environmental Policy. Pages 865-881 in: McKenzie, D.H., D.E. Hyatt & V.J. McDonald (eds.). *Proceedings of the International Symposium on Ecological Indicators*, held Oct. 16-18, 1990 in Fort Lauderdale, Florida. Elsevier Applied Science, London & New York.

Sinnige, C.A.M., W.L.M. Tamis & F. Klijn, 1992. *Indeling van bodemfauna in ecologische soortengroepen.* CML report 80, Leiden. 74 pp. met 8 bijlagen.

Witte, J.P.M., F. Klijn, F.A.M. Claessen, C.L.G. Groen & R. van der Meijden, 1992. A Model to Predict and Assess the Impacts of Hydrological Changes on Terrestrial Ecosystems, and its use on a Climate Scenario. *Wetlands Ecology and Management* 2(1992)/ 1/2: 69-83

Klijn, F., 1993. Alternatieven voor dijkverzwaring. Pages 143-168 in: J. Bervaes et al., 1993. *Landschap als geheugen; opstellen tegen de dijkverzwaring.* Uitgeverij Cadans Amsterdam.

Klijn, F., W.L.M. Tamis & C.A.M. Sinnige, 1993. Soil in ecosystems or ecosystems in soil? Classification of spatially nested ecosystems for ecosystem response modelling. Pages 102-104 in: Eijsackers, H.J.P. & T. Hamers (eds.). *Integrated Soil and Sediment Research: A Basis for Proper Protection.* Kluwer Academic Publishers, Dordrecht/ Boston/ London.

Runhaar, J. & F. Klijn, 1993. *Aanzet tot een aquatische ecoserie-indeling.* CML report 98, Leiden. 69 pp.

Claessen, F.A.M., F. Klijn, J.G. Nienhuis & J.P.M. Witte, 1994. Ecosystem classification and hydro-ecological modelling for national water management. Pages 199-222 in: F. Klijn (ed.). *Ecosystem classification for environmental management.* Kluwer Academic Publishers, Dordrecht/ Boston/ London.

Klijn, F., 1994. Spatially nested ecosystems, guidelines for classification from a hierarchical perspective. Pages 85-116 in F. Klijn (ed.). *Ecosystem classification for environmental management.* Kluwer Academic Publishers, Dordrecht/ Boston/ London.

Klijn, F. (ed.), 1994. *Ecosystem classification for environmental management.* Kluwer Academic Publishers, Dordrecht/ Boston/ London. 293 pp.

Klijn, F. & H.A. Udo de Haes, 1994. A hierarchical approach to ecosystems and its implications for ecological land classification. *Landscape Ecology* 9/2: 89-104

Sinnige, C.A.M., W.L.M. Tamis & F. Klijn, 1994. *Eigenschappen van ecosystemen voor ecotoxicologische effectvoorspelling.* CML report 104, Leiden. 99 pp. met bijlagen.

Udo de Haes, H.A. & F. Klijn, 1994. Environmental policy and ecosystem classification. Pages 1-21 in F. Klijn (ed.). *Ecosystem classification for environmental management.* Kluwer Academic Publishers, Dordrecht/ Boston/ London.

Gorree, M., J. Runhaar & F. Klijn, 1995. Stikstofbalansen voor natuurgerichte normstelling nutriënten. *Landschap* 12(1995)/2: 5-17

Klijn, F., R.W. de Waal & J. H. Oude Voshaar, 1995. Ecoregions and ecodistricts: ecological regionalizations for the Netherlands' environmental policy. *Environmental Management* 19(1995)/6: 797-813

Klijn, F. & R. Pastoors, 1995. Een landsdekkend overzicht van kwel: definitie- en schaalproblemen in de ecohydrologie. *Stromingen* 1(1995)/1: 29-44

Traas, T., F. Klijn & T. Aldenberg, 1995. *Ecosysteemgerichte risicobeoordeling van stoffen. Naar een model ter bepaling van het ecologisch rendement van brongericht stoffenbeleid.* RIVM rapport nr. 719102038, Bilthoven/ CML report 107, Leiden.

Claessen, F.A.M., G. Beugelink, J.P.M. Witte & F. Klijn, 1996. Predicting species loss and gain caused by alterations in Dutch national water management. European Water Pollution Control 6(1996)/5: 36-42.

Harms, H.B. & F. Klijn, 1996. Nederland in hokjes; de Landschapsecologische Kartering van Nederland. *Landschap* 13(1996)/4: ...-...

Klijn, F., C.L.G. Groen & J.P.M. Witte, 1996. Ecoseries for potential site mapping, an example from the Netherlands. *Landscape and Urban Planning* 35(1996): 53-70

Klijn, F., J. Kros & J. Runhaar, 1996. *Toekomstverkenning dosis-effectmodule DEMNAT: mogelijke toepassing van SMART in toekomstige versies van DEMNAT.* CML paper 21, Leiden.

Klijn, F., J. Runhaar & M. van 't Zelfde, 1996. *Ecoseries 2.1. Verbetering en operationalisatie van een classificatie van ecoseries voor DEMNAT 2.1.* CML report 121, Leiden/ DEMNAT 2.1 rapport nr. 2, RIZA, Lelystad.

Van Ek, R, F. Klijn, J. Runhaar, R. Stuurman, W.L.M. Tamis & J. Reckman, 1996. *Gewenste grondwatersituatie Noord-Brabant.* Deelrapport 1: Methode-ontwikkeling voor het bepalen van de optimale grondwatersituatie voor natuur. RIZA werkdocument ..., Lelystad.

Van Ek, R., J.P.M. Witte, J. Runhaar, F. Klijn, J.G. Nienhuis & J. Hoogeveen, 1996. *Beschrijving van het ecohydrologisch model DEMNAT versie 2.1.* DEMNAT 2.1 rapport nr. 1, RIZA, Lelystad.

Van Ek, R., J.P.M. Witte, J. Runhaar, F. Klijn, J.G. Nienhuis & U. Pakes, 1996. DEMNAT 2.1, latest innovations on a national dose-effect model for the analysis of dehydration of wetlands in the Netherlands. pp. 358-365 in Holzmann, H & H.P. Nachtnebel (eds.), 1996. *HydroGIS '96, International Conference on Application of Geographic Information Systemes in Hydrology and Water Resources Management, Vienna, Austria, April 16-19, 1996.* Volume of Poster Papers.

Van der Voet, E., F. Klijn, W.L.M. Tamis & R. Huele, in prep. *Regulatiefuncties van de biosfeer; aanzet tot een operationalisatie van de life supportfuncties van de biosfeer, toegespitst op de rol van soortenrijkdom.* Publicatiereeks VROM-SVS nr. .., Den Haag.