NATIONAL WATER MANAGEMENT AND THE VALUE OF NATURE

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NATIONAL WATER MANAGEMENT AND THE VALUE OF NATURE

Proefschrift
ter verkrijging van de graad van doctor
op gezag van de rector magnificus
van de Landbouwuniversiteit Wageningen,
dr. C. M. Karssen,
in het openbaar te verdedigen
op donderdag 9 april 1998
des namiddags te vier uur in de Aula

Het onderzoek in dit proefschrift werd verwezenlijkt met de financiële steun van het Rijksinstituut voor Zoetwaterbeheer en Afvalwaterbehandeling (RIZA) te Lelystad en het Rijksinstituut voor Volksgezondheid en Milieuhygiëne (RIVM) te Bilthoven.

Het RIZA bekostigde tevens de in deze publicatie opgenomen kleurenfiguren.

Druk: Offsetdrukkerij Van der Perk BV, Nieuw-Lekkerland
Omslag: Karin Luttik (ontwerp), Jan-Philip Witte (foto)

Nederlandse vertaling titel:

Nationaal waterbeheer en de waarde van natuur

CIP-DATA KONINKLIJKE BIBLIOTHEEK, DEN HAAG

Witte, Jan-Philip Maria

National water management and the value of nature. Jan-Philip Maria Witte Doctoral Thesis Wageningen Agricultural University. - With ref. - With summary in Dutch.
ISBN 90-5485-831-1

Stellingen

1. Wanneer men de vegetatie van Nederland wil beschrijven met behulp van het florabestand FLORBASE (Van der Meijden *et al.*, 1996), dan kunnen de ecologische soortengroepen uit het Leidse ecotopensysteem (Runhaar *et al.*, 1987) daarvoor beter worden gebruikt dan de plantensociologische soortengroepen van Westhoff & Den Held (1969).

   Runhaar, J., Groen, C. L. G., Van der Meijden, R., & Stevers, R. A. M., 1987. Een nieuwe indeling in ecologische groepen binnen de Nederlandse flora. Gorteria, 13(11/12):276-359.

   Van der Meijden, R., Groen, C. L. G., Vermeulen, J. J., Peterbroers, T., Van 't Zelfde, M., & Witte, J. P. M., 1996. Eindrapport FLORBASE-1. Rijksherbarium/CML, Leiden.

   Westhoff, V. & Den Held, A. J., 1969. Plantengemeenschappen in Nederland. Thieme & Cie, Zutphen.

   Dit proefschrift

2. Technische factoren kunnen van grote invloed zijn op de kwaliteit van biologische inventarisatiegegevens. Zo zijn moerasvegetaties in de eerste helft van deze eeuw waarschijnlijk slecht geïnventariseerd door het ontbreken van voor drassige terreinen bestemd schoeisel.

   Dit proefschrift

3. Inventarisatie-effecten van florabestanden kunnen ten dele door hiaatopvulling worden gecorrigeerd.

   Dit proefschrift

4. Het subjectieve oordeel van deskundigen over de kwaliteit van vegetaties, kan wiskundig goed worden nagebootst.

   Dit proefschrift

5. Het is verantwoord de waardering van classificatie-eenheden (soorten, vegetatietypen, ecosysteemtypen) uitsluitend te baseren op het criterium 'zeldzaamheid'.

   Dit proefschrift

6. In tegenstelling tot wat gebruikelijk is in waarderingsmethoden (bijv. Gremmen *et al.*, 1990; Mennema, 1973; Wheeler, 1988), dienen algemene soorten als Merel en Pinksterbloem zodanig te worden gewaardeerd, dat ze in een nationale boekhouding van natuurwaarden het grootste gewicht krijgen.

   Gremmen, N. J. M., Reijnen, M. J. S. M., Wiertz, J., & Van Wirdum, G., 1990. A model to predict and assess the effects of groundwater withdrawal on the vegetation in the Pleistocene areas of the Netherlands. Journal of Environmental Management, 31:143-155.

   Mennema, J., 1973. Een vegetatiewaardering van het stroomdallandschap van het Merkske (N.-Br.), gebaseerd op een floristische inventarisatie. Gorteria, 6(10/11):75-127.

   Wheeler, B. D., 1988. Species richness, species rarity and conservation evaluation of rich-fen vegetation in lowland England and Wales. Journal of Applied Ecology, 25:331-353.

   Dit proefschrift

7. De doelsoorten die de Rijksoverheid wil gebruiken voor de evaluatie van het natuurbeleid (Bal *et al.*, 1995), zijn aan de hand van dubieuze criteria vastgesteld.

   Bal, D., Beije, H. M., Hoogeveen, Y. R., Jansen, S. R. J., & Van der Reest, P. J., 1995. Handboek natuurdoeltypen in Nederland. IKC, Wageningen.

   Dit proefschrift

8. Grondwatermodellen zijn doorgaans niet in staat uitkomsten te produceren die aansluiten op de gevoeligheid van de vegetatie voor hydrologische veranderingen.

   Dit proefschrift

9. Gebieden hebben minder last van verdroging naarmate ze beter van Rijnwater kunnen worden voorzien.

   Witte, J. P. M. Stromingen 2(1):52.

10. Het natuuronderzoek in Nederland wordt niet alleen beperkt door een gebrek aan veldgegevens, maar ook door de geringe bereidheid van ecologen om die gegevens af te staan.

11. De AMOEBE –een grafische presentatievorm waarmee de toestand van ecosystemen wordt weergegeven (Ten Brink & Hosper, 1989)– herkent men aan het vrijwel ontbreken van herkenbare kenmerken.

    Ten Brink, B. J. E., & Hosper, S. H., 1989. Naar toetsbare ecologische doelstellingen voor het waterbeheer: de AMOEBE-benadering. H2O, 20:612-617.

12. Wanneer men een mens 'een beest' noemt, dan doelt men gewoonlijk op heel menselijke eigenschappen; een beest wordt echter 'net een mens' genoemd wanneer deze dierlijke gedragingen vertoont.

13. Kijkt iemand overdag eerst op zijn horloge voor antwoord te geven op de vraag wat hij wil drinken, dan heeft hij vast een drankprobleem.

    Observatie Herman Witte, Heemstede

14. Door overwegend spectaculaire foto's te presenteren –van bijv. travestieten, aidslijders, bandieten, zenuwzieken in het voormalige Oostblok– wekt de organisatie van de grote tweejaarlijkse tentoonstelling van Naarden de indruk, dat ze fotografie als vorm van beeldende kunst nauwelijks serieus neemt.

15. Bij personeelsreducties aan de universiteiten dient de kwaliteit van het onderwijs en onderzoek richtinggevend te zijn en niet –zoals nu het geval is– de leeftijd van het personeel.

16. Het onderwijs in Nederland heeft behoefte aan een minister met een gebrek aan ambities en ideeën.

Stellingen behorend bij het proefschrift *National Water Management and the Value of Nature* van J. P. M. Witte, Wageningen, 9 april 1998.

Abstract

Witte, J.P.M., 1998. *National Water Management and the Value of Nature.* Doctoral thesis, Wageningen Agricultural University, the Netherlands. 223 pp., 103 figs., 28 tables, 5 appendices.

In order to attune its water management to the demands of nature, the Dutch Government uses the ecohydrological DEMNAT for the analysis of scenarios. The input to this model consists of changes in hydrological variables that may be computed with present hydrological models. The output consists of changes in the botanical quality of various ecosystem types, and of resulting values for nature conservation. The applied ecosystem types are defined on the basis of abiotic factors that determine the plant species composition of a vegetation. In the Netherlands these are: salinity, moisture regime, nutrient availability, acidity. Water management measures may cause changes in these four factors and, as a result, change the species composition of a vegetation.

Species of the Dutch flora are allotted to the ecosystem types. It is questioned whether the resulting ecological groups are appropriate for describing the plant cover of the Netherlands with the aid of FLORBASE. This national database contains data, per kilometer square, on the presence of indigenous plant species. To answer the question, a comparison is made with a division of species into phytosociological groups by the Dutch standard work of Westhoff & Den Held (1969). On the basis of a correlation analysis, it is concluded that the ecological groups are of better use for the analysis of the national plant cover than the phytosociological groups.

The ecological species groups are used in combination with FLORBASE to make nation-wide maps of ecosystem types. On the basis of both the number and the indicative value of diagnostic species, the botanical quality of each kilometer square is assessed for each ecosystem map. The boundaries of the quality classes are obtained by expert judgement. It is possible, however, to compute class boundaries with a mathematical procedure, also for grid cells larger than 1 km^2. The maps are corrected for regional differences in the intensity of the plant inventories.

It is argued that the conservation value of classification units - such as species and ecosystem types - is especially related to the criterion 'rarity'. The measuring of rarity is discussed and a mathematical formula is presented for the valuation of classification units. Eight methods for the botanical valuation of areas are compared. Those methods which are based on species richness and species conservation values yield results that experts find unsatisfactory. The most favored method, however, is based on the quality classes of the ecosystem maps in combination with the conservation values of the ecosystem types.

Additional index words: national water management, ecohydrology, ecosystem classification, vegetation mapping, conservation valuation, flora databases.

Voor Zuster Paula

Voorwoord

Eind jaren tachtig werkte ik in Arnhem bij het Rijksinstituut voor Integraal Zoetwaterbeheer en Afvalwaterbehandeling (RIZA). Deze onderzoeksafdeling van Rijkswaterstaat was bezig met analyses en berekeningen voor de Derde Nota Waterhuishouding. In de Eerste Nota Waterhuishouding van 1968 werd met geen woord gerept over de natuur. In de Tweede Nota (1985) stonden mooie woorden naast een kleurenfoto van een Orchidee. Maar in de Derde Nota Waterhuishouding - die eind 1989 zou verschijnen - moest de natuur de aandacht krijgen die zij verdiende. Daartoe werd een projectgroep gevormd, onder leiding van Frans Claessen. Deze projectgroep - waaraan ik mocht deelnemen - ging onderzoeken wat de gevolgen zijn van het nationale waterbeheer en waterbeleid voor de terrestrische natuur in Nederland.

Om iets zinnigs te kunnen zeggen over zulke gevolgen, moet je natuurlijk eerst geschikte informatie bezitten over het voorkomen van de natuur in Nederland. Deze informatie bleek echter niet voorhanden te zijn. Daarom togen Frans Claessen en ik in 1988 naar het Rijksherbarium te Leiden, want dat bezat belangwekkende gegevens: landelijke kaartjes met de verspreiding van wilde planten. Deze kaartjes waren ook nog eens handig opgeslagen in een computerbestand, het zogenaamde ATLAS-bestand. We maakten toen voor het eerst kennis met Ruud van der Meijden van de afdeling Nederlandse flora. De samenwerking tussen Ruud en mij leidde in 1990 tot een publicatie over ecosysteemkaarten: van het ATLAS-bestand afgeleide kaarten met daarop de verspreiding van ecosysteemtypen in Nederland.

Deze ecosysteemkaarten gebruikte ik vervolgens als invoer voor een computermodel (DEMNAT) waarmee voorspeld kan worden hoe ecosysteemtypen veranderen door ingrepen in de waterhuishouding van Nederland. In dit model werd ook de deskundige kennis op ecologisch gebied verwerkt van Frans Klijn, Kees Groen en Han Runhaar, toen nog allen werkzaam bij het Centrum voor Milieukunde in Leiden (CML).

Het model DEMNAT bleek een onuitputtelijke bron te zijn voor leuk en nuttig onderzoek. Zo kreeg het Rijksinstituut voor Volksgezondheid en Milieuhygiëne (RIVM) begin jaren negentig behoefte aan een ecohydrologisch voorspellingsmodel. Door het RIVM moesten namelijk analyses worden uitgevoerd ten behoeve van het Beleidsplan Drink- en Indutriewatervoorziening. Het oog viel op DEMNAT. Een samenwerkingsverband werd gesmeed tussen RIVM, RIZA, Rijksherbarium, CML en de vakgroep Waterhuishouding van de Landbouwuniversiteit - waar ik inmiddels werkte. Deze groep ging DEMNAT eens grondig oppoetsen: geografische gegevens verfijnen en toevoegen, relaties aangaande het functioneren van ecosystemen aanpassen aan de laatste gegevens en inzichten, herschrijven van de computerprogramma's door professionele programmeurs. Ook na het verschijnen van het Beleidsplan - in 1993 - bracht dezelfde groep nog diverse verbeteringen aan, onder meer voor de Vierde Nota Waterhuishouding (1997). Mijn bijdragen aan de nieuwe modelversies van DEMNAT - uit de periode 1991 – 1997 - vormen de bouwstenen voor dit proefschrift.

Nu ik kort de achtergronden van het onderzoek heb beschreven, wil ik nog een aantal mensen noemen aan wie ik veel te danken heb.

Een van hen is de al eerder genoemde Ruud van der Meijden, mijn co-promotor, met wie ik boeiende discussies over floristiek, vegetatiekunde en waanzinnige wetenschap kon voeren. Het enthousiasme waarmee hij me al die tijd heeft aangespoord is zeer waardevol geweest. Verscheidene publicaties hebben we samen geschreven, waarvan de weerslag vooral te vinden is in Hoofdstuk 4 van dit proefschrift.

Mijn promotor Reinder Feddes heeft mij de afgelopen jaren op gezette tijden vaderlijk toegesproken: dat ik toch echt eens met schrijven moest beginnen. Deze zachte maar onverbiddelijke psychologische druk heeft geholpen. Zijn commentaar op de concept-stukken was punctueel en ter zake.

Veel genoegen heb ik beleefd aan de contacten met de jongens van het CML: Kees Groen, Frans Klijn en Han Runhaar. Samen vormden we een onderzoeksclub die de kennis en gegevens voor DEMNAT genereerde. Ik heb veel van ze geleerd en genoten van de heftige discussies die ik met ze voerde. Onze samenwerking leidde tot verscheidene publicaties, die vooral in Hoofdstuk 2 zijn verwerkt. In dit hoofdstuk komt ook de bijdrage van Remco van Ek (RIZA) aan de orde, die de afgelopen jaren aan de onderzoeksclub is toegevoegd. Sinds 1994 is hij op inhoudelijk gebied de coördinator van het DEMNAT-onderzoek. De toewijding en het inzicht waarmee hij dat doet verdienen navolging.

Frans Claessen (RIZA) vroeg als voorzitter van de DEMNAT-vergaderingen vaak het onmogelijke van de onderzoekers. Met zijn vriendelijke maar volhardende aandrang wist hij het nog te bemachtigen ook. In de DEMNAT-geschiedenis is er een periode geweest waarin een buitenstaander zich op slinkse wijze meester wilde maken van het onderzoek. Frans doorzag de piraat en wist het gevaar af te slaan.

Guus Beugelink (RIVM) vormde als secretaris het cement tussen diegenen die betrokken waren bij DEMNAT. Zijn optreden was altijd genuanceerd, vriendelijk en doelgericht.

Gerard Nienhuis (RIVM) verrichtte ondersteuning bij het verwerken van de immense hoeveelheid geografische gegevens. Jippe Hoogeveen (RIZA) en Geert van Wirdum (IBN-DLO) leverden waardevolle bijdragen aan het onderzoek dat leidde tot Hoofdstuk 3. Samen met Hank Vermulst (RIZA) en Remco van Ek voorzag Jippe mij bovendien van de gegevens die nodig waren voor Hoofdstuk 2. Ronald

Lieste, Som Gan (beiden RIVM), Ubo Pakes, Herbert Bos (beiden RIZA) en Bert Bleij (LUW) schreven computerprogramma's voor DEMNAT.

Ernestiene Scheer uit Oosterbeek corrigeerde mijn Engelse teksten en vertaalde enkele hoofdstukken. Ze deed dat met begrip voor de inhoud van de tekst en - voor zover ik dat kan beoordelen - met veel gevoel voor taal.

Zonder het florabestand FLORBASE zou dit proefschrift nooit geschreven kunnen zijn. Ik ben daarom veel dank verschuldigd aan zowel de vele instanties als de particulieren (zie Appendix A voor een overzicht) die de gegevens van FLORBASE verzamelden.

Wageningen, november 1997

Contents

Chapter 1

General introduction

1.1 Problem definition

The Netherlands - a flat country, measuring no more than 35,000 km^2 - is almost entirely situated within the estuarine area of the rivers Rhine, Meuse and Scheldt. About 25% of its surface is below mean sea level and, if it had not been for the dikes and dunes, 65% of the country would be flooded at high levels of see and rivers (Colenbrander *et al.*, 1989). Roughly spoken, the country may be divided in a relatively low and a relatively high part (Fig. 1.1): the West and the East

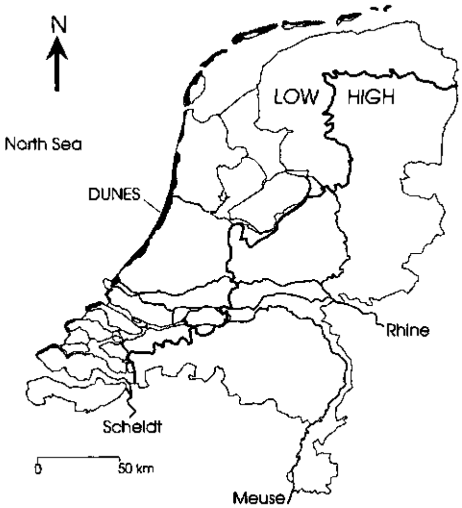

Figure 1.1: Division of the Netherlands in a western and northern 'low' part, and an eastern and southern 'high' part (after: Colenbrander *et al.*, 1989). One meter above mean sea level (m.s.l.) was chosen as the critical value for the boundary between the two parts (bold line). In the 'low' part, only the dunes along the North Sea coast are higher than 1 m m.s.l..

are low, whereas the North and the South are 'high'. The low part, which is mainly covered by Holocene clay and peat, is predominately constituted of so-called 'polders'. A polder is a piece of land, made by man, in which the water level is artificially controlled. It is constructed by isolating land from the surrounding hydrological regime by means of dikes, upon which superfluous water is pumped out. In former times this was done by means of windmills. The high part, on the other hand, mainly consists of Pleistocene cover sands. Here, there are also wet areas (for instance in the brook valleys), but the groundwater level is generally situated much lower. It is hardly controlled, since drainage is mostly brought about by gravity.

Due to the low level of the land, groundwater tables are very shallow in the Netherlands. For instance, in 90% of the country the water table is less than 1 m below surface in winter and less than 2.5 m below surface in summer (Colenbrander *et al.*, 1989). As a consequence, a considerable part of the indigenous plant life is characteristic for wet to moist conditions. This may be illustrated by the fact that about 40% of the plant species and more than half of the vegetation types are exclusively or largely bound to the influence of groundwater (Londo, 1988; Colenbrander *et al.*, 1989). Therefore, many conservation values in the Netherlands are associated with wet and moist ecosystems, such as fens, bogs, dune slacks, wet heathlands, swamp woodlands and wet meadows.

The Netherlands are heavily industrialized and densely populated (446 inhabitants per km^2; Centraal Bureau voor de Statistiek, 1992), which inevitably results in a strong claim to land: 95.4% of the surface area is used for economic or urban purposes so that no more than 4.6% of 'waste land' is left for nature (Centraal Bureau voor de Statistiek, 1992). Many animal and plant species are to a considerable extent restricted to small reserves (often measuring less than 1 km^2) or spots within an agricultural landscape, so that they are strongly dependent on the water management of the surroundings.

Especially in the second half of this century the water regime of most areas changed drastically by human interference. Since 1950 several hundreds of land improvement plans were realized, often including a radical revision of the water management. Moreover, groundwater extraction for industrial and drinking water purposes increased from 0.3×10^9 m^3a^{-1} in 1955 up to 1.1×10^9 m^3a^{-1} in 1990 (Beugelink *et al.*, 1992). As a result, groundwater levels dropped (Fig. 1.2). For instance, in Pleistocene areas that were subjected to land reconsolidation, the groundwater table dropped by 35 cm on average (Rolf, 1989). To compensate for the negative effects of low groundwater levels on agriculture, surface water from the rivers Rhine and Meuse is divided over the land (Ministerie van Verkeer en Waterstaat, 1989). This water is rich in nutrients (which for a substantial part comes from intensive agriculture) and has a chemical composition that is often different from that of the local surface water (Roelofs, 1989; Koerselman & Verhoeven, 1993).

As a result of the man-induced 'drought', ecosystems suffered severely. A nationwide investigation in the loss of conservation values since 1950 showed moderate

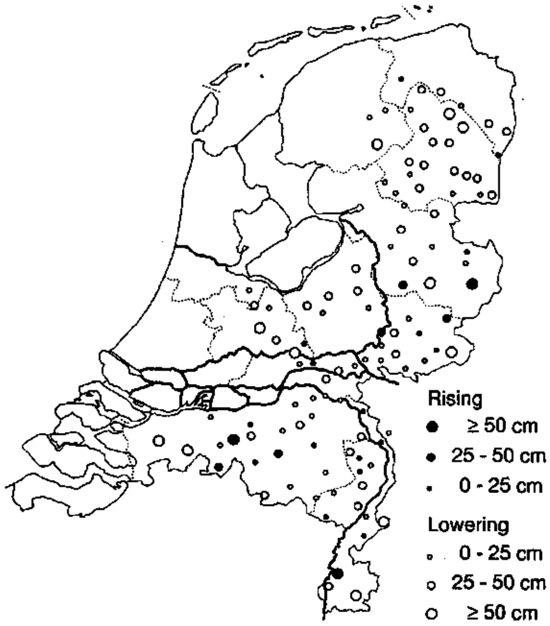

Figure 1.2: Change in the groundwater table depth over the period 1950-1986 (after: Rolf, 1989).

to severe damage to ecosystems in about 50% of the groundwater-dependent nature reserves (Fig. 1.3) (Van Amstel *et al.*, 1989; Runhaar *et al.*, 1996a). Especially ecosystems of nutrient-poor sites were affected (Fig. 1.4). In Dutch this major environmental problem is called 'verdroging', to be translated as *desiccation*. Verdroging (desiccation) may be defined as the decline of nature due to falling water tables, decreasing intensities of upward seepage, and the inlet of allochtonous surface water to the extent that it is used to compensate the negative effects of groundwater shortages (Van Amstel *et al.*, 1989). More information on this complex environmental problem will be given in Section 1.5.3.

In former times the water management of the Netherlands was mainly attuned to economic interests, such as shipping and agriculture, as well as to the safety of the Dutch (protection against flooding). However, nature having become a scarce commodity, water management now also aims at the conservation, restoration and creation of both nature reserves and natural elements in the rural environment. For instance, in 1990 the Dutch Parliament ordered that the area of desiccated nature should be restored by at least 25% before the turn of this century. Since that time, several 'rewetting' projects have been carried out, each of them subsidized with a 50% by the National Government (De Vrieze & Pellenbarg, 1994; Runhaar *et al.*, 1997a). Moreover, the Government tries to reduce the extraction of groundwater: the industries have to increase the amount of surface water for cooling purposes in order to save valuable clean groundwater. Also the drink-

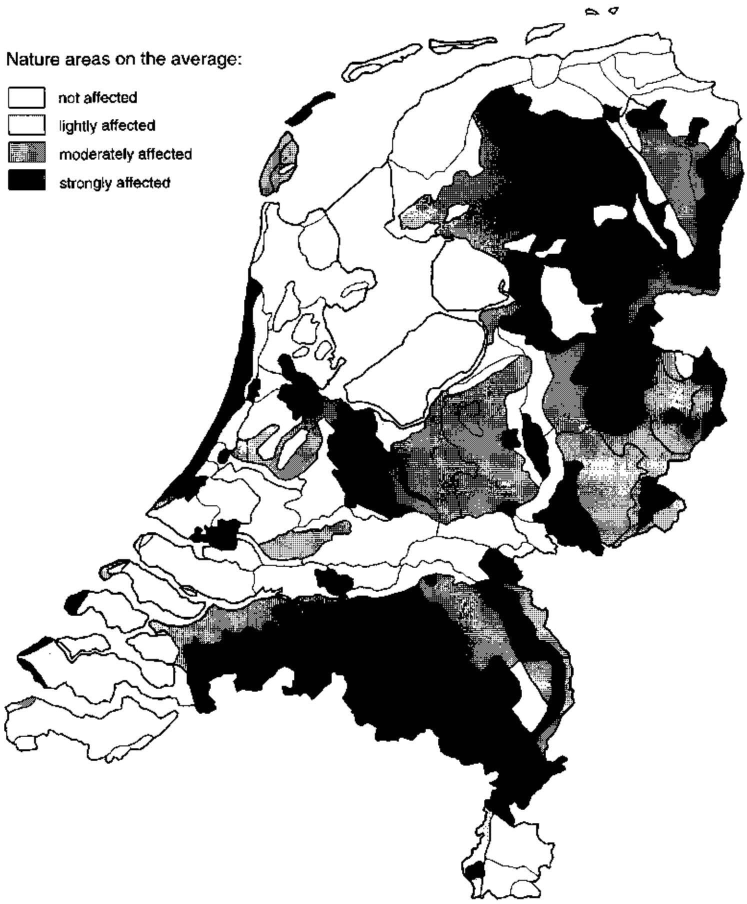

Figure 1.3: Deterioration of nature reserves as a result of desiccation (after: Van Amstel *et al.*, 1989).

ing water companies have to convert to surface water (Ministerie van Verkeer en Waterstaat, 1997a; Ministerie van Volkshuisvesting, Ruimtelijke Ordening en Milieubeheer, 1995). In dry periods many provincial authorities even put a complete ban on sprinkler irrigation. All in all, many efforts are taken to restore nature.

When making policies for the quantitative and qualitative water management of the Netherlands, the Dutch Government uses various kinds of computer models, e.g. for the calculation of the groundwater levels, the quality of the surface waters, or the water depth of the main shipping routes. These models are used to analyze the consequences of interference in water management for various land use functions, such as agriculture, shipping, industry and drinking water supply. In this way, promising water management options are selected for defining sound policies or for a detailed analysis. In recent policy documents on water management much attention is paid to the deterioration of nature (Ministerie van Verkeer en Waterstaat, 1989; Ministerie van Verkeer en Waterstaat, 1993; Mini-

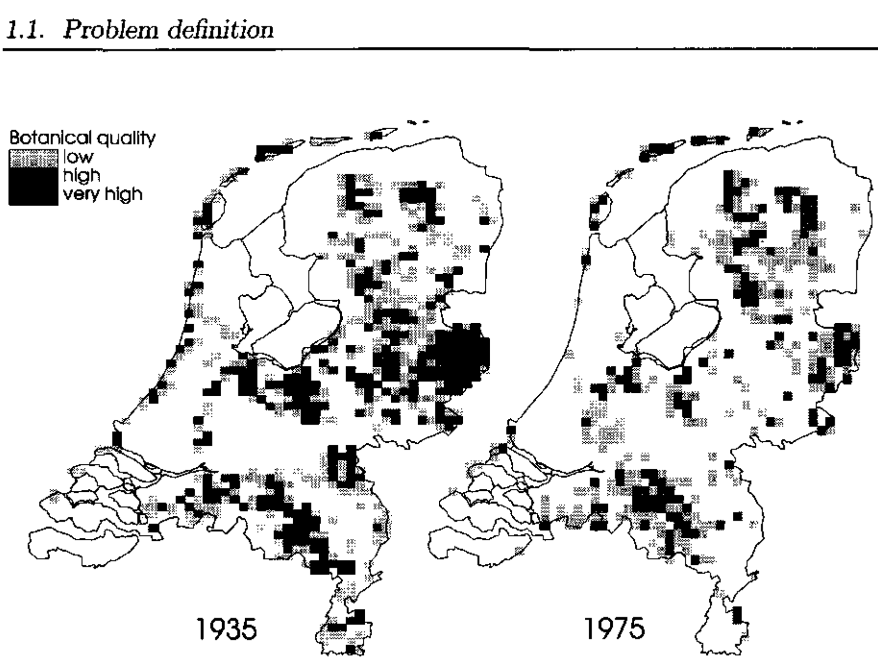

Figure 1.4: Changes in the botanical quality of ecosystems of wet, nutrient-poor and weakly acid soils over approximately the period 1935-1970 (after: Witte & Van der Meijden, 1990).

sterie van Verkeer en Waterstaat, 1997a; Ministerie van Verkeer en Waterstaat, 1997b; Ministerie van Volkshuisvesting, Ruimtelijke Ordening en Milieubeheer, 1993; Rijksinstituut voor Volksgezondheid en Milieuhygiëne, 1993). In order to give the natural environment a prominent place in the analyses for these documents, the Dutch Government - in close corporation with a number of research institutes - developed the national ecohydrological model DEMNAT (Dose Effect Model for terrestrial NATure).

DEMNAT facilitates the evaluation of the impacts of water management scenarios on nature. It uses ecosystem types for its computations, whereas most ecohydrological models make computations for plant species. Plant species have the advantage of being clearly distinguishable units that, unlike ecosystem types, are independent of a - to a greater or lesser degree - arbitrarily chosen typology. However, the use of ecosystem types in a national model has several advantages, as will be shown in the course of this publication.

In DEMNAT, only the vegetation is considered as representative for the biotic part of the ecosystem. There are two reasons for this approach: firstly, the relationship between the abiotic environment and plant life is quite direct, whereas the relationship between the abiotic environment and fauna is more indirect. And secondly, the value (expressed as a number) of an area for nature protection is usually deduced from the vegetation (Van Wirdum, 1986).

1.2 Aim of this publication

One of the major problems dealt with in the making of DEMNAT was the lack of suitable geographical information on ecosystems in the Netherlands. At the start of the DEMNAT project - in 1989 - the most appropriate information on a national scale was a 1 : 250,000 map with "nature reserves, woodlands and wetlands" (Bakker *et al.*, 1989). This map shows units like 'woods and shrubs', 'wet and dry heath' and 'swamp'. These units were considered too rough and by no means adapted to problems concerning water management. As a consequence, it was decided to make use of the records of plant species from the National State Herbarium (see Section 1.4). Questions dealt with in this publication are:

- What are the effects of water management measures on the vegetation and the nature conservation value of the Netherlands (Chapter 2)?
- Can flora records be used to derive national distribution maps of ecosystem types, susceptible to water management (Chapter 4)?
- Are these ecosystem types suitable for describing the vegetation of the Netherlands, or should one use a more traditional ('phytosociological') approach for this purpose (Chapter 3)?
- What is the value of groundwater-dependent ecosystem types and areas in terms of nature conservation (Chapter 5)?

1.3 Botanical terminology

Clarification of terminology should facilitate the understanding of the contents of this publication. Unfortunately, in ecology there is a proliferation of terms, as well as of their interpretation. This implies that it is inevitable that my descriptions - extracted from several sources (Mueller-Dombois & Ellenberg, 1974; Schaminée *et al.*, 1995; Schroevers, 1982; Shimwell, 1971; Tansley, 1935; Westhoff & Den Held, 1969) - sometimes clash with the opinions of others.

Species that settled spontaneously - i.e. that grow in the wild - are called *wild* species. A list of wild species of a certain area is called the *flora* of that area. So we may speak of the flora of the Netherlands, the flora of a certain nature reserve and the flora of a grid-cell. The range of distribution of a particular species on earth is called the *area* of that species.

Any plant cover consisting of wild species will be called a *vegetation.* This implies that a crop or an afforestation is not regarded as a vegetation. One might however consider all the weeds growing between the cultivated species to form a vegetation because these weeds have established unintentionally. A *plant community* is that part of the plant cover that consists of wild species which typically occur together in repeating groups of associated plants. As in the Netherlands, this term might be associated with the French-Swiss school (see below), I prefer to use the term *vegetation* also for such actual pieces of the plant cover.

The vegetation of a certain area can be classified in *community types* or in *vegetation types*. Usually this is done on the basis of differences and similarities in the composition of species. Vegetation types are used to characterize and map the vegetation of an area. They have no absolute reality in nature, they are "somewhat like arithmetic means and ranges that may give lesser or closer fit to a population of numbers" (Mueller-Dombois & Ellenberg, 1974). I will apply this term to any classification system, on any hierarchial level of vegetation types. A species with a marked preference for a certain vegetation type may feature as a *diagnostic species*: a species by which an actual vegetation (plant community) in the field may be recognized as a member of a particular vegetation type.

By nature, not all vegetation types are equally rich in plant species. For instance, a bog vegetation, is relatively poor in species, whereas a quagfen vegetation (NL: trilveenvegetatie) has a high species richness. An important concept in this publication is *completeness*, which denotes the 'saturation' of a vegetation with species characteristic for the vegetation type in question. I will discuss this concept thoroughly in Chapter 4, but for the time being 'completeness' is defined as 'relative species richness' or as 'botanical quality'.

Numerous methods are available to describe the vegetation on the basis of a classification in types. In the Netherlands, as in many other European countries, the method of the *French-Swiss* school is generally applied. This method uses *relevés* (*relevé* is French for abstract) to investigate the vegetation: sample stands in which all the species are recorded, including their abundances. The French-Swiss school also goes by the name of *phytosociology* (Braun-Blanquet, 1932), which reflects that the plants in a community have some influence upon one another and/or that they have something in common with their environment. A brief description of the French-Swiss school is given in Appendix C.

Every plant species makes specific demands on its environment, e.g. on air, temperature, light, and the availability of nutrients and water. Species have different demands. A species like *Urtica dioica* (UK: Common Nettle; NL: Grote brandnetel) prefers nitrogenous soils, ranging from wet to dry, whereas for instance *Rhynchospora alba* (UK: White Beak-sedge; NL: Witte snavelbies) markedly prefers nutrient-poor and wet circumstances. The common demands of individuals belonging to a certain species is called the *habitat* of that species. So the habitat of *Urtica dioica* is characterized by nitrogenous soils. Compared with 'habitat' - which is also used in ecology for animal organisms - the term *site* seems more appropriate for the environment of plant life. In this study, the term 'site' will not only be applied to plant species, but also to vegetation types. Furthermore, depending on the context the word 'site' might refer to a specific location where the demands of a certain species or a vegetation type are met. Thus, 'site' may be used in an abstract as well as in a concrete sense. In case of any risk of misunderstanding, I will use 'site type' for the abstract sense.

In the above mentioned examples, only the optimum environment of species was given. The term *ecological amplitude* is used for the range of environmental values, in the sphere of which a species occurs. Some species have a marked preference for a narrow range of an environmental factor, others are less choosy. A species with a narrow ecological amplitude may be used as a kind of measuring device for the location where it grows. Such a species is called an *indicator species*. So e.g. *Rhynchospora alba* is an indicator species for nutrient-poor and wet circumstances. An *ecological species group* is a list of indicator species with a common preference for a certain habitat. As will be shown in this publication, it is possible to map a vegetation by using ecological species groups. In that case the indicator species of the group in question features as a diagnostic species in vegetation mapping.

Tansley (1935) introduced the term *ecosystem* for communities with their habitats, expressing the view that organisms should not be considered in separation from their specific environments. Both organism and environment form a functional system in nature (an ecosystem). As vegetation and site together form an ecosystem, in practice terms like 'ecological group', 'vegetation type' and 'site type' might alternate in one and the same text, depending on the context and on what the author wishes to emphasize. When for instance ecological species groups are used in vegetation mapping, the result might be referred to as 'vegetation map', 'ecological group map' or 'ecosystem map'. In the first case (vegetation map) the description of the plant cover is stressed, in the second case (ecological group map) the emphasize is on the occurrence of indicator species. In this publication I prefer to use the third case (ecosystem map) because I want to stress both plant cover and the functional relationship between the vegetation and its habitat.

1.4 National databases of the Dutch flora

1.4.1 History

In this publication several national flora databases are analyzed. On a national grid they contain information about the presence of indigenous species that grow in the wild. They do not contain detailed information *within* a grid-cell, for instance about the number and cover of species or about the joint occurrence of species within a vegetation. Only vascular species are considered because other taxonomical groups, such as mosses and lichens, have been investigated less thoroughly.

The awareness that facts about nature are a necessity in environmental and nature conservation policies, undoubtedly stimulated the inventory of the Dutch flora already in the beginning of this century. In 1902 amateur botanists began a systematic survey on the basis of the topographical maps, which at that time measured 30 × 20.83 km (Smit & Verschoof, 1980). Species observations were recorded in grid-cells of 1.25 × 1.04 km or, in case of incidental and poorly located observations, in 5 × 4.17 km cells. The survey ended in 1949 when the size of the topographical maps changed to 30 × 25 km.

From 1950 onwards, species were first recorded on a 5 × 5 km national grid. Later on, in the seventies, several provincial authorities started professional inventories on a very detailed scale (measuring less than 1 km^2). As to the method and intensity with which the inventories were carried out, there were considerable differences between the provinces (Groen *et al.*, 1992). Some provinces, like Utrecht (see Fig. 4.1, Section 4.1, for the location of the Dutch provinces), investigated the flora in 'landscape elements'. Others, like Gelderland, used vegetation relevés. At that time, also amateur botanists began to investigate on a more detailed scale, viz in 1 km squares (*km-squares*). Since 1988 the FLORON foundation has coordinated and stimulated the work of private amateurs and regional working groups (Groen *et al.*, 1995).

1.4.2 Databases

In 1978, with the rise of the computer, the ATLAS-project started, aiming at a national database with flora records on a 5 × 4.17 km national grid for the inventory period 1902–1950 (ATLAS_I) and on a 5×5 km grid for the observations from 1950 – 1987 (ATLAS_II). The cells of 5 × 4.17 and 5 × 5 km are referred to as *hour squares* (Dutch: 'uurhokken'), since it takes ca. one hour to cross a cell on foot. For ATLAS_I the greater part of the observations took place around 1935, for ATLAS_II this is 1975 (personal communication Van der Meijden). The records were published in an atlas of three volumes. The first appeared in 1980 and contained very rare and extinct species (Mennema *et al.*, 1980), the second one was published in 1985 with rare species (Mennema *et al.*, 1985) and on completion of the ATLAS project in 1989, the third volume with common species was issued (Van der Meijden *et al.*, 1989). ATLAS_I contains 433,286 records, all collected by amateurs. In ATLAS_II - with 495,119 records - the data from the first two volumes were collected by amateurs, whereas in the third volume, also records of the provincial surveys are included.

With the ATLAS database, distribution maps were made of the botanical quality of ecosystems in the Netherlands (Fig. 1.4) (Van der Meijden *et al.*, 1989; Witte & Van der Meijden, 1989; Witte & Van der Meijden, 1990). The maps from ATLAS_II served as input to DEMNAT-1, the first version of the model (Witte, 1990; Witte *et al.*, 1992). However, the application of these maps in DEMNAT is to some extent questionable. Shortcomings are that: (1) the grid-cells of 5 × 5 km are large, (2) the inventory period of three decades (1950 – 1980) is relatively long and might result in the combination of species which did not really occur at the same time, and (3) the ATLAS_II records are rather outdated.

The experiences with DEMNAT-1, together with the increasing facilities of computers and the growing concern about the natural environment, stimulated the Dutch Government to finance the development of FLORBASE, a database with more recent information on a kilometer square basis (Fig. 1.5). In the first version of FLORBASE, records from 22 sources were incorporated, most of which were readily accessible, such as the provincial databases and the database of the FLORON foundation (Groen *et al.*, 1992). For the second version (FLORBASE-1), new, often smaller databases were explored and analogous information in literature was digitized (Van der Meijden *et al.*, 1996).

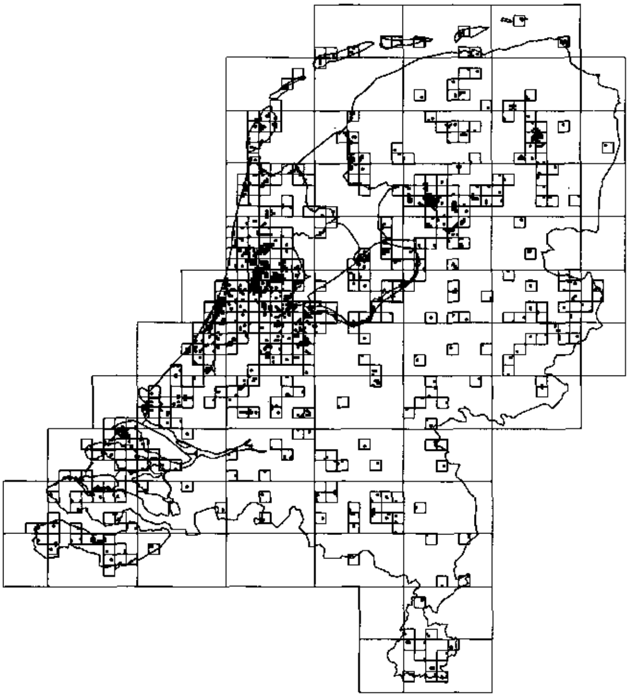

Figure 1.5: Distribution of *Dactylorhiza majalis* (NL: Brede orchis; UK: Western Marsh-orchid) according to FLORBASE-2c. Also its distribution in hour squares (of 5×5 km) is shown as well as the contours of the current topographical maps. This species is spotted in 1346 km-squares and 473 hour squares. On the basis of the latter number and Table 1.1 this species would be ascribed to UFK7.

In this publication *FLORBASE-2c (released in November 1997) is used for the making of up-to-date ecosystem maps.* It contains 6.4 million records from the period 1975-1995; $\frac{2}{3}$ of which was supplied by professional biologists and $\frac{1}{3}$ by amateurs. The amateurs take a special interest in rare species, which is illustrated by the fact that they gathered half of the records on those species. Color Fig. 4.24 gives the number of species per km-square in FLORBASE-2c. For some regions, such as the provinces of Limburg, Brabant and Overijssel, the numbers are low but fortunately they primarily consist of indicator species.

1.4.3 Selection of species for this publication

The Dutch flora comprises nearly 1500 wild vascular plant species. However, not all of these species can be used for the analyses in this publication. As I want to use species as indicators for their natural environment, only species that grow in the wild are applicable. These species may be selected from the 'Standard List of the Dutch flora 1990' (Van der Meijden *et al.*, 1991), in which all wild species are listed. A further selection is needed for the following two reasons.

Firstly, there are species and subspecies that were hard to identify during the inventory periods (*Alchemilla, Callitriche*). Therefore, a number of taxonomically allied species are assembled to species groups if the separate species share the same ecological indication. Seven *Alchemilla*-species, for instance, are united as 0025 *Alchemilla* ('spec.').

Secondly, most of the tree and shrub species are left out. It is common knowledge that, in the Netherlands, most individuals from this category were planted. However, field observers never agreed on the question whether a planted tree should be recorded or not. The exclusion of the tree and shrub species implies that, in the following chapters, ecosystems of woods and shrubs will be identified by means of herb species only.

All in all, ca. 250 species from the Standard List are omitted in the analyses presented in this publication.

1.4.4 The *UFK*-scale for species

For a number of purposes it is desirable to have access to information about the degree to which a plant is rare or common in the Netherlands. This information can be obtained from the national flora databases. The so-called *UFK*-scale by Van der Maarel (1971), based on the number hour squares in which a certain species occurs in the Netherlands (Table 1.1, Fig. 1.5), is generally applied in the Netherlands. The abbreviation *UFK* stands for the Dutch 'UurhokFrequentieKlasse', to be translated as Hour square Frequency Class. The *UFK*-scale runs from UFK1 (extremely rare) to UFK9 (extremely common). In several editions of the Standard List of the Dutch flora the numerousness of species expressed on this scale was published. The next to last edition (Van der Meijden *et al.*, 1991)

Table 1.1: Division in 'Uurhok Frequentie Klassen' (*UFK*'s) according to Van der Maarel (1971). With this division a *UFK* can be ascribed to a species on the basis of the number of hour squares in which that species occurs. For an example, see Fig. 1.5.

UFK	Number of hour squares
1	1 - 3
2	4 - 10
3	11 - 29
4	30 - 79
5	80 - 189
6	190 - 410
7	411 - 710
8	711 - 1210
9	$\geq$ 1211

gives an estimation of the UFK's in 1940 and in 1990, based on flora databases as well as on expert judgement.

1.5 Hydrology and the requirements of plant species

1.5.1 Hydrology and site

A plant is linked to its environment by means of its roots, stem, leaves and reproductive organs. Direct factors that determine plant life are the availability of light, water, carbon, phosphorus, nitrogen, micro-nutrients, etc.. In practice, these *physiological* factors are unsuitable for describing the site of a species because they are difficult, if not impossible, to measure. Besides, they are too variable in time (Runhaar & Udo de Haes, 1994).

It was Van Wirdum who introduced the adjectives *operational* and *conditional* in Dutch ecohydrology (Van Wirdum, 1979). Just like Runhaar & Udo de Haes (1994) I will apply the term 'operational' to measurable factors in the close vicinity of the plant that largely determine the physiological conditions in which the plant lives. In literature there is much agreement on the fact that *salinity, moisture regime*, *nutrient availability* and *acidity* are the most relevant operational soil factors for plant life (Braun-Blanquet, 1932; De Boer, 1984; De Vries, 1948; Ellenberg, 1991; Etherington, 1982; Klapp, 1965; Landolt, 1977; Londo, 1988; Van Wirdum & Van Dam, 1984). These factors, in their turn, are determined by conditional factors, such as soil type and seepage intensity. I will use the adjectives 'operational' and 'conditional' in a relative sense in order to distinguish between relatively direct factors and relatively indirect factors. From physiological via operational to conditional there is a decreasing causal relation with plant life. In general, the site of a species should be described on the basis of operational factors. As stated, physiological factors are hard to measure, whereas conditional factors are too indirect to be generally applicable.

An example of an important conditional factor is upward seepage, which regulates the moisture regime, nutrient availability and acidity of the soil to a considerable extent (Grootjans, 1985). In some areas with poor, sandy soils, upward seepage with calcium-rich and bicarbonate-rich groundwater is a prerequisite for a mesotrophic or an oligotrophic environment. In other areas such an environment may be caused by a high buffering capacity of the soil or the influence of calcium-rich and bicarbonate-rich surface water (Van Wirdum, 1991). I think it is doubtful whether generally applicable seepage indicators really exist because the main influence of upward seepage is probably indirect (conditional). Perhaps *Hottonia palustris* (UK: Water-violet; NL: Waterviolier; see Fig. 1.6) is a reliable indicator of seepage in the Netherlands. The distribution of this species correlates very well with areas of upward seepage (Van Moorsel & Barendregt, 1993). A possible explanation for this phenomenon is that *Hottonia* only consumes carbon in the

Figure 1.6: A ditch with the seepage indicator *Hottonia palustris* in full bloom.

form of CO_2, so that it is mainly found in ditches that are fed by CO_2-rich seepage water (De Lyon & Roelofs, 1986). Another explanation is that this species responds to the constant temperature of the seepage water. Both explanations concern the response of *Hottonia* to a direct factor, i.e. to an *operational* factor.

An example of a factor that has both an operational and a conditional effect is the depth of the groundwater table. Some species are adapted to survive in wet places that are saturated with water for at least part of the year. Permanently or periodically waterlogged sites are characterized by anaerobic conditions in the root zone. The low redox-potential often results in the formation of potentially toxic substances, such as Fe^{2+}, Mn^{2+} and H_2S. Species growing in wet sites are often characterized by air-space tissues, transporting oxygen to their roots, or by a very shallow root system that hardly penetrates the soil. *Sphagnum* species even have no root system at all. Deep-rooting species often have an ability to oxygenate the soil around their roots, so that potentially toxic, reduced substances are oxidized and made harmless. On the basis of these physiological considerations it is not surprising that there is a relatively strong relationship between the groundwater level and the fraction of 'wet species' in the vegetation (Fig. 1.7): on wet sites the groundwater level directly influences the amount of oxygen in the root zone, so that it may be considered as an operational factor.

In dry places, where periods of moisture deficits occur, other physiological adaptations are needed. Perennial species are sometimes capable of storing water (succulents), while many species have an extensive root system. A sclerophylic anatomy helps to prevent structural deformation due to wilting. Annual species avoid water stress by germinating and growing in periods of sufficient water supply (Etherington, 1982). The formation of an open vegetation reduces the loss of water by evapotranspiration. Species that lack adaptations for both wet and dry circumstances need 'moist' sites, i.e. sites that are neither too wet nor too dry.

The greater the depth of the groundwater table, the more indirect its effect on

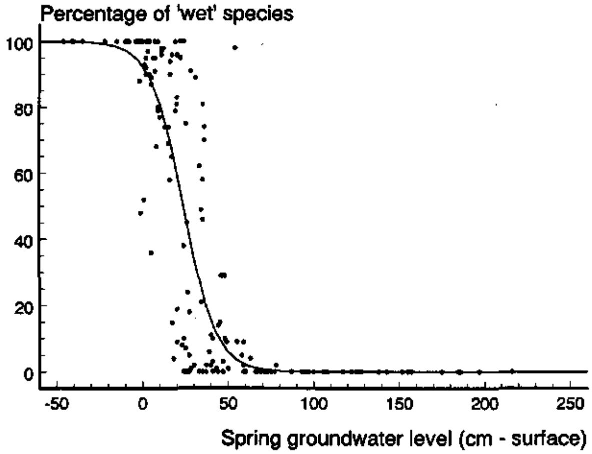

Figure 1.7: Relation between the average groundwater level in spring (1980-1986) and the percentage of 'wet species' in the vegetation (adapted from Runhaar *et al.*, 1997d). Based on 202 relevés (Runhaar, 1989b).

the plants. In moist and dry circumstances, differences in species composition of the vegetation relate to the availability of moisture for evapotranspiration. 'Moist species' need enough water for evapotranspiration at a potential rate. 'Dry species' are physiologically adapted to periods of drought and are capable of reducing their evapotranspiration rate. The amount of water for evapotranspiration depends on the upward capillary transport of water from the groundwater table to the root zone - *the capillary rise* - as well as on the amount of moisture that a vegetation can extract from the root zone itself - *the amount of available soil moisture* (Miller & Donahue, 1990). Capillary rise, in its turn, is not only dependent on the depth of the groundwater table, but also on the texture of the soil. In comparison with a coarse sand soil for instance, the capillary rise in a loamy soil is more intense, causing - with similar groundwater depths - the fraction of 'moist species' in the vegetation to be larger. The amount of available soil moisture also depends on the soil texture. Species of moist sites may grow on places with a very low groundwater table if there is much available soil moisture. According to my own observations *Fraxinus excelsior* (UK: Ash; NL: Gewone es) is such a species, growing on loess soils on hill tops, whereas in most places in the rest of the Netherlands the occurrence of this species is linked to the influence of the groundwater table.

Hence, for 'moist species', the groundwater table is a conditional factor that, together with soil texture and rooting depth, indirectly regulates the availability of soil moisture. Bearing this in mind, it is obvious that the relation between groundwater depth and the fraction of 'dry species' in the vegetation is not very distinct (Fig. 1.8A). The soil moisture deficit, on the other hand, seems to be a more general (operational) measure for vegetations of dry sites (Fig. 1.8B).

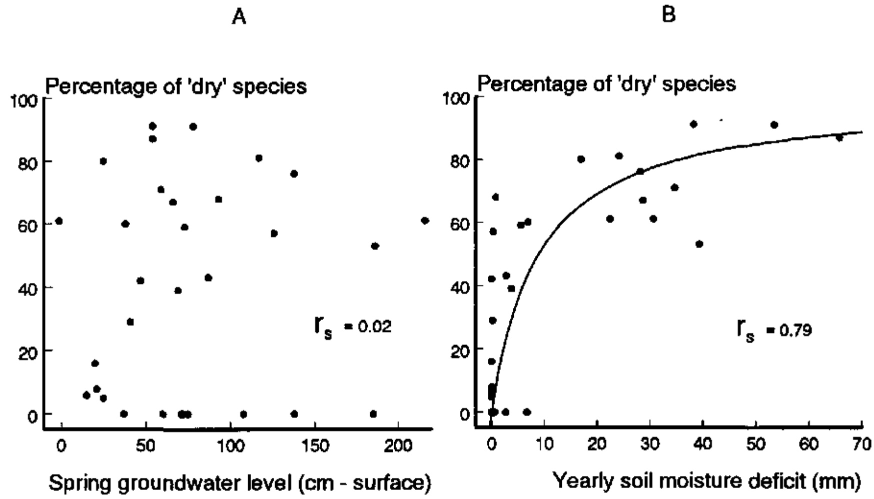

Figure 1.8: Percentage of 'dry' species in the vegetation versus: A - the measured average spring groundwater level; B - the computed annual soil moisture deficit (1980-1987). Based on 32 relevés, most of which are influenced by the groundwater table (adapted from Runhaar *et al.*, 1997d). r_s: Spearman rank correlation coefficient (with ties) (Sachs, 1982, p. 401).

1.5.2 Hydrology and landscape

Hydrology determines the differentiation in site conditions and, consequently, the species composition in a landscape, as was stressed by several authors (Both & Van Wirdum, 1981; Beltman & Grootjans, 1986; Dijkema *et al.*, 1985; Kemmers, 1986; Wassen, 1990; Van Wirdum, 1991). As the relation between hydrology and species composition on a landscape scale is not very direct, it is hard to deal with this relation without stating commonplaces. Therefore, the importance of hydrology will only be discussed very briefly in this section.

Water is a major factor in soil development. Some soils were deposited by water, as is obvious with the marine and fluviatile clay soils in the lower parts of the Netherlands. Topographical differences in altitude underlie flow patterns, recharge and discharge of groundwater. When there is no artificial water supply, recharge areas (downward seepage) only receive water by precipitation. As a result of a precipitation excess, nutrients are washed out and, with that, a nutrient-poor soil will develop. Moreover, when the buffering capacity of the substratum is low, the soil will become acid as a result of the protons that are released by mineralization. The chemical composition of the groundwater in recharge areas strongly resembles rainwater, in Van Wirdum's terminology (1991) it is *atmotrophic*, which can easily be demonstrated by a low electric conductivity and a low Ionic Ratio ([Ca]/([Ca] + [Cl]), where concentrations are in meq l^{-1}; Van Wirdum, 1980). Groundwater flows from recharge areas, where the hydraulic heads are high, to discharge areas (upward seepage), where they are low. It is self-evident that discharge areas have longer periods with high groundwater levels than recharge areas. During its flow from recharge to discharge areas, the groundwater will predominantly be enriched by calcium and bicarbonate and become *lithotrophic* (Van Wirdum, 1991), with a high Ionic Ratio. Some species and vegetation types 'need' lithotrophic water to survive. In some low polders in the western part of the Netherlands, upward seepage of brackish groundwater may occur. As mentioned in the previous paragraph (Section 1.5.1), salinity is an important site factor for the species composition of the vegetation.

Of course, surface water also plays an important role, not only because the surface water level controls the flow of groundwater, but also because of the relatively quick transport of nutrients and other elements by surface water. In valleys of brooks and rivers inundation is an event of great ecological importance; in fens the chemical quality of the inlet-water largely determines what type of vegetation and what species will appear.

1.5.3 Ecological effects of hydrological changes

As the vegetation of the Netherlands is very dependent on groundwater and surface water, it is susceptible to hydrological changes. Especially in the second half of this century, the water regime of many areas changed radically, mainly by an increasing extraction of groundwater, drainage of agricultural land and the inlet of surface water from the rivers Rhine and Meuse. On many locations this resulted in a decreasing groundwater level, a decreasing intensity of upward seepage and an increasing influence of Rhine-water. Fig. 1.9 shows in a very simplified way how these hydrological changes may adversely influence important operational factors of wet and moist ecosystems. This figure has been compiled from various sources (Barendregt, 1993; Beltman & Grootjans, 1986; Etherington, 1982; Grootjans, 1985; Koerselman, 1989; Koerselman & Verhoeven, 1993; De Mars, 1996; Roelofs, 1989; Runhaar & Van 't Zelfde, 1996; Van Wirdum & Van Dam, 1984; Vermeer, 1985).

A falling groundwater level may lead to shortages in the water supply to the vegetation and, as a result, in a *physiological* desiccation of the vegetation: species that are adapted to wet and moist environments will disappear. A groundwater fall may also cause an increased aeration which, in turn, promotes mineralization and, consequently, eutrophication. Hence, species characteristic for nutrient-poor sites will disappear. When organic matter is mineralized, protons are released and acidification of the soil takes place, causing species of neutral and alkaline sites to vanish.

The availability of phosphor is largely regulated by adsorption on calciumhydroxydes (at $pH > 6.5$) and ironhydroxydes ($pH < 6.5$) (Stumm & Morgan, 1981). In this adsorbated form it is not available for plants. Hence, an influx of calcium or iron-rich water by upward seepage may lead to the development of mesotrophic and oligotrophic sites. Moreover, calcium and bicarbonate in upward seepage water form an important buffer against acidification by percolating rainwater.

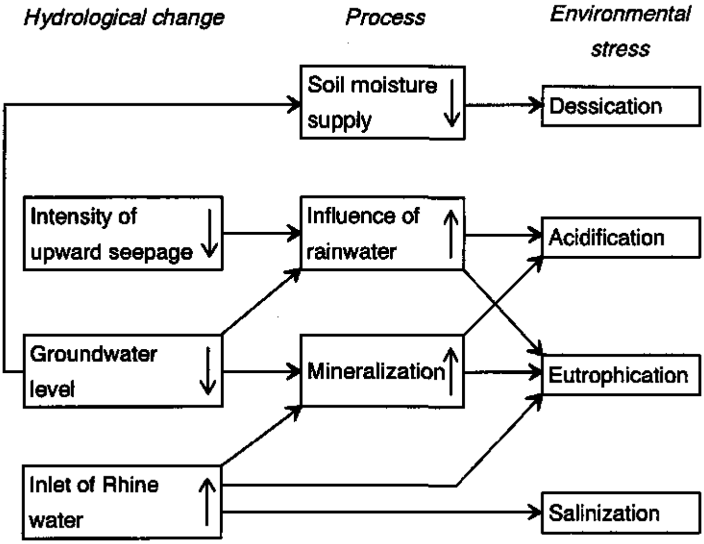

Figure 1.9: Main effects of hydrological changes on important operational factors of wet and moist sites. Direction of change indicted by arrows: (↑) increase, (↓) decrease.

Hence, when the soil is originally influenced by lithotrophic upward seepage, a decreasing groundwater level may enhance both eutrophication and acidification. Of course, both effects may also take place when the intensity of upward seepage diminishes.

The inlet of surface water may lead to eutrophication, especially of aquatic ecosystems, since this water - in many cases from the rivers Rhine and Meuse - is often rich in phosphor and nitrogen. Even when nutrient concentrations are low the inlet-water may stimulate mineralization, leading to 'internal eutrophication'. An explanation for this phenomenon is that the inlet-water often has a higher *pH* than the original water. Moreover the high sulfate concentrations may contribute to internal eutrophication, since sulfur is capable of forming a complex with iron, leaving less for the fixation of phosphor (Caraco *et al.*, 1989; Koerselman & Verhoeven, 1995). Finally, a higher salinity of the inlet-water will have marked effects on the vegetation: species of fresh water will disappear.

As stated, the representation in Fig. 1.9 is highly simplified; some processes even may have positive effects. A groundwater level fall, for instance, may also reduce the availability of phosphor, because in its oxidative state iron (Fe^{3+}) is more capable of fixating phosphor than in its reduced state (Fe^{2+}) (Patrick & Khalid, 1974). Nevertheless, Fig. 1.9 gives a good overall expression of how hydrological changes may generate several environmental stressors. It also shows that there is a close relation between these stressors. Hence, what is referred to by ecologists as "the desiccation of nature" (NL: verdroging) is more than solely physiological desiccation, i.e. the shrivelling of plants as a result of water shortage. Also eutrophication, acidification and even salinization (when surface water is used to counteract low groundwater levels) stem from desiccation. In fact, the influence of these stressors on the vanishing of plant species is thought to be of more importance than the influence of physiological desiccation (Gremmen, 1990). The strong connection between the environmental stessors also explains why, generally speaking, it is impossible to judge the desiccation of the vegetation solely from its species composition. A dominance by grasses in a heathland for instance, may originate from a fall in the groundwater level, but may also be caused by an increased atmospheric deposition of nitrogen.

The connection between the environmental stressors is also shown in Fig. 1.10. In this figure the decline in the magnitude of various wet terrestrial ecosystems over approximately the period 1935 – 1975 is plotted against the susceptibility of these ecosystems to a lowering of the groundwater level. Fig. 1.10 shows that the decline of ecosystems over the period 1935 – 1975 is to a considerable extent correlated with the susceptibility to a groundwater lowering. Though a conclusion on the basis of a correlative relation is not without risk, one might think that the lowering of the groundwater level is probably the main cause of the decline of ecosystems. However, a closer look at Fig. 1.10 reveals that the decline also corresponds to the factor 'nutrient-richness': ecosystems that are very susceptible to a groundwater lowering, are ecosystems of nutrient-poor sites (and - vice versa - relatively insusceptible ecosystems are those of very nutrient-rich sites). As nutrient-poor ecosystems are susceptible to eutrophication, one might as well

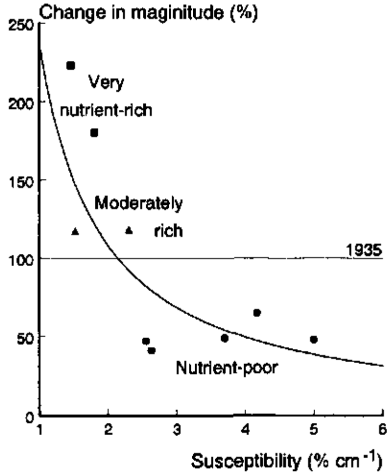

Figure 1.10: Relation between the susceptibility of wet terrestrial ecosystems to a lowering of the spring groundwater table and the decline in the magnitude of these ecosystems over approximately the period 1935 – 1975 (Witte & Van der Meijden, 1990, susceptibility adapted to Arts *et al.*, 1997). Susceptibility defined as the average decline (in terms of percentage) of the botanical quality per cm lowering of the spring groundwater table.

conclude from Fig. 1.10 that the decline of ecosystems is caused by an increased input of nitrogen which, in its turn, is due to a raised atmospheric N-deposition or to excessive manuring. The environmental stressors appear to be closely related, indeed.

1.6 The ecotope system

1.6.1 The need for indicator values of plants

The relation between the distribution of plant species and physical site factors, such as groundwater level, soil-*pH* and N-content, has been the subject of many studies (Barendregt, 1993; De Lange, 1972; De Lyon & Roelofs, 1986; Koerselman, 1989; Kruijne *et al.*, 1967; Noest, 1991; Runhaar, 1989a; Vermeer, 1985). However, these studies have not yet resulted in a list of the requirements of all of the Dutch plant species, expressed in physical factors. Such a list is not to be expected in the near future either, since the study of the physical requirements of species is hampered by several problems.

- It is difficult to carry out experiments with a natural plant cover. In agriculture it is possible to study the effect of a single factor, simply by manipulating this particular factor and keeping all the other factors constant. In nature, this approach is often impossible because site factors usually are closely linked and hard to manipulate separately. A change in the groundwater level for instance, does not only result in a different moisture supply, but also influences the soil-*pH* and the availability of nutrients (Fig. 1.9).
- A plant does not immediately disappear when its environment becomes unsuitable. Especially in the case of geophytes and species with a vegetative reproduction, it may take a long time before they disappear. Because of this phenomenon of a retarded response, a plant may falsely be associated with site measurements.
- Some species are mainly found in places with a very strong heterogeneity of site factors, such as soil-*pH*. These places are difficult to measure. Measurements taken 'somewhere in the vicinity' of the roots may result in a strong correlative relation, while the underlying causal mechanisms may be very weak or even absent.
- In nature, as apposed to agriculture, species grow in a heterogeneous plant cover, consisting of many different, interacting species. Results from pot experiments on a single species are of limited value because the response of a species to a site factor in the homogeneous pot may differ considerably from its response in the wild.
- Nature conservation often takes a special interest in rare species. Random sampling is necessary when the ecologist wants to back up his conclusions, but in doing so he misses species that are the most desired (i.e. rare). It should be noted that the majority of species is very rare in a statistical sense.

Fortunately, several experts combined information from their field observations, literature, measurements and experiments into a judgement - published in a list - about the requirements of species on an ordinal or a nominal scale. The advantage of this approach is that it allows experts to compensate for statistical oversampling, weight differences in results stemming from differences in measuring methods, filter out apparent correlations, etc..

In the Netherlands the list of Dutch *phreatophytes* of Londo (1975, 1988) is generally known. Phreatophytes are plants that occur exclusively in or are largely limited to the sphere of influence of the groundwater table. In Londo's nominal list, species are put into categories that reflect the connection with the groundwater table. With the remarks in Section 1.5.1 about the conditional mechanism of the water table in mind, it is quite obvious that Londo distinguishes soil-dependent categories (for instance: 'lime aphreatophytes', 'dune phreatophytes') and that he advises to calibrate his list for each local application separately.

Another indicator list that is generally accepted is the one from Ellenberg (1979, 1991), who placed the ecological optima of species on ordinal scales of site factors like 'salinity', 'moisture regime', 'nutrient availability' and 'acidity'. His list is meant for the western part of Middle Europe, especially the area between northern edge of the German low mountain range and the central Alps, but in practice it is also used in other regions, such as the Netherlands. The explanation for this wide use is that his list deals with operational factors.

Especially developed for the Netherlands is the *ecotope system* (Runhaar *et al.*, 1987; Runhaar & Udo de Haes, 1994; Stevers *et al.*, 1987), which is meant for environmental impact assessment. Unlike other indicator lists, this system ascribes species to ecosystem types instead of separate site factors. Another difference is that information about the ecological amplitude of species is taken into account. The ecotope system, described below, is used in this publication.

1.6.2 The classification of ecotope types

The ecotope system consists of a classification of ecosystem types and an assignment of the Dutch plant species to ecological species groups. The basic unit is the *ecotope*, defined as: "a spatial unit that is homogenous in vegetation structure, stage of succession and in the dominant abiotic factors that determine the species composition of the vegetation" (Stevers *et al.*, 1987). Hence, from the biotic part of the ecosystem only the vegetation is taken into account. As the vegetation is described in combination with its habitat, an ecotope is an ecosystem: an ecosystem of a certain size (small) and a certain homogeneity (Runhaar & Udo de Haes, 1994).

For the classification of ecotope types, abiotic and biotic factors that determine the species composition of the vegetation have been used. As the classification had to be generally applicable, operational site factors were chosen as abiotic classification characteristics. The main abiotic characteristics of the ecotope system are the important factors (Section 1.5.1) 'salinity', 'moisture regime' (characterizing

Table 1.2: Classification characteristics and classes of the ecotope system.

Characteristic	Classes [symbol]
Salinity	Fresh [-] Brakish [b] Saline [z]
Vegetation structure	Terrestrializing [V] Water [W] Pioneer [P] Grassland [G] Tall herbaceous [R] Woods and shrubs [H]
Moisture regime	Water [1] Wet [2] Moist [4] Dry [6]
Nutrient availability	Low [-] Moderate [7] High [8] Moderate to high [9]
Acidity	Acid [1] Neutral (Weakly acid) [2] Alkaline [3]

both the water regime and the oxygen regime), 'nutrient availability' and 'acidity'. Fig. 1.9 clarifies why the ecotope system is very suitable for the evaluation of water management measures: precisely these factors are susceptible to changes in the water regime.

Climate is also an important factor that influences the vegetation both directly (e.g. through frost) and indirectly (e.g. through soil development). However, it has not been used as a classification characteristic because within the Netherlands climate differences are small. Vegetation structure was chosen as the main biotic classification characteristic. Vegetation structure might be interpreted as indicative for the operational factors 'time' and 'vegetation management' (Runhaar & Udo de Haes, 1994; Runhaar *et al.*, 1987; Stevers *et al.*, 1987).

For each classification characteristic several classes have been distinguished, each of which is indicated by a symbol (Table 1.2). Subsequently, ecotope types have been constituted by combining classes, resulting in ecotope types such as G27: a grassland (G) on a wet (2), moderately nutrient-rich (7) soil. A description of the classification classes is given in Appendix B, including a brief comparison with the indicator scales of Ellenberg (1979, 1991) and Klapp (1965). The class 'grassland' for instance, is described as a low vegetation with herbs, bryophytes and/or dwarf shrubs. So the contents of this class does not correspond to the general concept of a grassland, being a low vegetation dominated by grasses.

Not all the theoretically possible combinations of classes have been distinguished as ecotope types (Table 1.3). Some combinations are ecologically irrelevant. For example, in ecosystems that are very rich in nutrients, the influence of acidity on the species composition is far less pronounced than in nutrient-poor ecosystems. Therefore, in very nutrient-rich ecosystems acidity has not been used as a classification characteristic. In addition, many combinations of classes do not occur in the Netherlands (for instance the combination 'woods and shrubs' - 'saline').

An (operational) site type within the ecotope system is defined by a combination of abiotic class characteristics. Hence, each 'cell' in Table 1.3 corresponds with a site type (example site type X27: wet, moderately nutrient-rich). On the basis of vegetation structure, each site type is subdivided in ecotope types.

The ecotope system also distinguishes additional characteristic classes, e.g. for dynamics and vegetation management. For national applications the characteristic

Table 1.3: Types distinguished in the ecotope classification. Within ecotope types P47 and G47 calcareous subtypes are distinguished (these are indicated by an asterisk). Example: the code ' G47*' denotes a grassland (G) on a moist (4), moderately nutrient-rich (7) and calcareous (*) soil.

	Fresh					Brackish	Saline
	Nutrient-poor			Moderately nutrient-rich	Very nutrient-rich		
	Acid	Neutral	Alkaline				
Water	V11	V12	W13	V17	V18	bV10	
	W11	W12		W17	W18	bW10	
Wet	P21	P22	P23	P27	P28	bP20	zP20
	G21	G22	G23	G27	G28	bG20	zG20
	H21	H22		R27	R28	bR20	zR20
				H27	H28		
Moist	P41	P42	P43	P47*	P48	bP40	
	G41	G42	G43	G47*	G48	bG40	
	H41	H42	H43	R47	R48	bR40	
				H47	H48		
Dry	P61	P62	P63	P67	P68	bP60	
	G61	G62	G63	G67	G68		
	H61	H62	H63	R67	R68		
				H69			

class 'calcareous' is worth mentioning. It is used to subdivide pioneer vegetations and grasslands of moist and moderately nutrient-rich soils into a calcareous type and a non-calcareous type (see the asterisks in Table 1.3).

1.6.3 Ecological species groups

The species composition of the ecotope types is described by means of ecological species groups (Runhaar *et al.*, 1987). An ecological species group comprises plant species that are characteristic for a certain ecotope type. Species that occur in two or more ecotope types, have been assigned to more than one ecological species group. In this way, the ecological amplitude of species is taken into account. By way of example, Table 1.4 lists the ecological species group of ecotope type P21 (pioneer vegetation on a wet, nutrient-poor and acid soil). Only vascular plant species are presented because these are relevant for this publication, but in reality also mosses and liverworts (Dirkse & Kruijsen, 1993) and *Characeae* (Van Raam & Maier, 1993) have been ascribed to the groups.

The initial assignment of species to ecological groups is based on expert judgement and on national and international literature, concerning for example indicator

Table 1.4: The ecological species group of ecotope type P21 (first column). Other ecotope types in which the species occur are presented in the second colum. After: Runhaar et al. (1987).

Species characteristic for ecotope type P21	Other ecotope types in which the species commonly occurs
Drosera intermedia	
Drosera rotundifolia	G21, G22
Juncus bulbosus	V11, V12, P22
Lycopodiella inundata	
Rhynchospora alba	
Rhynchospora fusca	

values of plant species (Clausman *et al.*, 1987; Ellenberg, 1979; Klapp, 1965; Londo, 1975, *fide* Runhaar & Udo de Haes, 1994). As a second step the consistency of the groups was tested using ca. 50,000 relevés from all over the Netherlands. The relevés served to check whether species attributed to a certain ecological species group actually occur in combination with other species from the same group (Runhaar *et al.*, 1987).

1.6.4 Critical comparison with the French-Swiss school

In the Netherlands it is common practice to describe the vegetation of an area according to the method of the French-Swiss school of phytosociology (Appendix C). In this publication, however, I will use ecological species groups for this purpose, because in this way a direct relation between vegetation and the abiotic environment is brought about.

Conceptually, there is a fundamental difference between the classification of ecotope types and that of phytosociological vegetation types (Runhaar & Udo de Haes, 1994) (Fig. 1.11). For the ecotope classification, a limited number of classification characteristics were applied, as well as pre-defined classification classes. Any ecotope type can be placed in this framework. The addition of new vegetation samples (relevé P in Fig. 1.11) will not lead to changes in the classification. New insights, however, may very well bring them about. Because of the fact that the delimitation of ecotope types is based on a hypothesis of how the vegetation is determined by site factors, the classification method might be referred to as *deductive* (Udo de Haes & Klijn, 1994). The classification of phytosociological vegetation types, on the other hand, is purely based on field data (relevés), and not on any concept of how nature functions. As the types may be regarded as nodes in a multi-dimensional continuum, relevés that are intermediate between types cannot always be classified. Additionally, the addition of new relevés may lead to changes in the classification (Runhaar & Udo de Haes, 1994). Udo de Haes and Klijn (1994) call this an *inductive* classification approach.

In practice, there are more differences. Even in recent publications (Everts *et al.*,

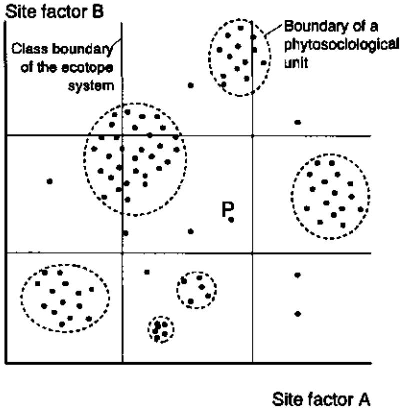

Figure 1.11: Comparison of an ecotope classification with fixed classes related to site factors A and B, with a vegetation classification based on the clustering of relevés. Each dot represents one relevé. After: Runhaar & Udo de Haes (1994).

1986; Schaminée *et al.*, 1995), some supporters of the phytosociological method suggest that species occur together in a vegetation because they overlap as to their ecological amplitude. Hence, the species composition of a vegetation would give us quite precise information about a *homogeneous* site. However, as Van Wirdum (1986) pointed out, species might also occur together in a vegetation, because each species finds there its own micro-habitat. In that case, the site of the vegetation is *heterogeneous*, i.e. consisting of different species micro-habitats. Van Wirdum (1991) convincingly showed that quagfens (NL: trilvenen), with their slightly acid hummocks and neutral to alkaline hollows, are heterogeneous.

The concept of a heterogeneous site sure is in support of the phytosociological method, since this method only uses differences and similarities in species composition of relevés to delimit community types, irrespective of underlying causal factors. The concept of a homogeneous site, on the other hand, is found in the definition of an ecotope (Section 1.6.2), as being "homogenous in ...the dominant abiotic factors that determine the species composition of the vegetation". This definition seems to leave little room for heterogeneous sites. However, as in the ecotope system 'homogeneity' is not explicitly defined, nor measured, in practice the difference with the phytosociological approach is less pronounced than it might seem. In Section 4.7, I will return to the problem of homogeneity.

In Dutch literature (Everts *et al.*, 1986; Grootjans *et al.*, 1987), the ecotope classification has been criticized for the fact that it uses fixed class boundaries to define its types. This would do no justice to the spatial groupings of species as they occur in nature, since such groupings would be split up by the classification or - just as bad - they would be merged (Fig. 1.11). Moreover, with its fixed boundary classes the ecotope classification takes not account of the fact that the ecological significance of site factors, like acidity and nutrient-availability, may be different for each kind of vegetation. In my opinion however, one should question how serious these drawbacks really are. Firstly, because in the ecotope system, only those abiotic site factors were used as classification characteristics, which determine the plant species composition of the vegetation. Secondly, because "many vegetations scientist will readily agree that the basic units they are classifying are really ecosystems" (Daubenmire, 1968, *fide* Runhaar & Udo de Haes, 1994): so relations with site factors always play an important (implicit) role in the classification of phytosociological vegetation types.

Moreover, it is good to realize that from a scientific viewpoint, also the phytosociological approach has its shortcomings (for a critical review, see Kershaw & Looney, 1985). For instance, the whole basis of the system is established on a markedly *non-random* sampling procedure. Intermediate situations - as reflected by point P in Fig. 1.11 - are ignored or rejected as being 'not homogeneous'. Kershaw & Looney even attribute the success of the French-Swiss school to prejudice: "Thus it should be clearly recognized that the success of the Braun-Blanquet method of establishing clear-cut associations rests entirely on the careful choice of the stand with the stipulations of minimal area and homogeneity" (Kershaw & Looney, 1985, p. 175). Another serious point of criticism has to do with the segregation of vegetation units from the 'synthesis table' (Appendix C). Poore (1955, *fide* Kershaw & Looney, 1984, p. 162) sums up this procedure as "muddled and haphazard", while Mueller-Dombois & Ellenberg (1974, p. 175) point at its subjective nature: "the solving of the question as to which of the units can be considered associations, depends on the personal judgement of the investigator".

As to the classification of the vegetation, I firmly agree with Mueller-Dombois & Ellenberg (1974, p. 29) who state that: "The absence of absolute boundaries between adjacent communities in the field is no greater a problem in community identification than is the absence of absolute boundaries between two adjacent but different soils in their identification. Sharp boundaries are rare between any related natural phenomena. Yet, soil classifications have received little objection on this account, while the concept of classifying vegetation is generally less well understood".

1.7 Outline of this publication

Chapter 2 is dedicated to a description of the ecohydrological model DEMNAT, emphasizing its general structure, the combination of various geographical data and the responses of ecosystems to hydrological changes. Also some other approaches in Dutch ecohydrological modelling will be discussed.

The ecotope system is intended to be used in environmental impact assessment, but is it also appropriate for describing the plant cover of the Netherlands? I will deal with this question in Chapter 3 by comparing the ecological species groups with the phytosociological species groups of Westhoff & Den Held (1969).

Chapter 4 describes a method for deriving distribution maps of ecosystem types from the flora records in FLORBASE. Much attention will be paid to the reliability and accuracy of the maps. A method will be presented to correct the ecosystem maps for regional differences in the intensity with which the plant inventories for FLORBASE were carried out.

With respect to nature conservation, the ecosystem types of Chapter 4 are not all equivalent. In Chapter 5 a system is presented for the assessment of a quantitative conservation value for the individual ecosystem types. Furthermore, a method is presented for the calculation of the conservation value of km-squares with the aid of flora records.

I tried to write each chapter in such a way, that it can be read independently from the other chapters. Accordingly, each chapter ends with an elaborate discussion. So if the reader is for instance particularly interested in classification systems or in the assessment of conservation values, he can confine to respectively Chapter 3 and Chapter 5 and skip the rest.

The nomenclature in this publication follows Van der Meijden (1996) for vascular plant species, Westhoff & Den Held (1969) for phytosociological plant communities, and Runhaar *et al.* (1987) for ecological species groups. All colored maps of one chapter are placed together at the end of that chapter.

Chapter 2

Ecohydrological modelling

2.1 Introduction

To illustrate the kind of problems that ecohydrological modelling deals with, I should like to introduce the following imaginary example: a nature reserve in a valley has a vegetation that is dependent on upward seepage of lithotrophic groundwater (with rare plant species like *Carex pulicaris* and *Cirsium dissectum*; UK: Flea sedge and Meadow thistle; NL: Vlozegge en Spaanse ruiter). Water infiltrating in an adjacent ice-pushed ridge flows through the subsoil to the direction of the valley, where it - eventually - will exfiltrate. During its transport the groundwater becomes enriched with Ca^{++} and HCO_3^-, while its pH rises. The upward seepage creates the wet, nutrient-poor and weakly acid site that is typical for the vegetation in this nature reserve. Suppose now, that a drinking water company takes up the plan to extract groundwater in the ice-pushed ridge. How may the effects of that measure on the vegetation be predicted and how should these effects be judged upon?

In Fig. 2.1 a relational scheme for an ecohydrological prediction is presented. It is a *general* scheme, based on the way ecohydrological modelling until now has been carried out *in practice*. Apart from discussing this scheme in general terms, I will also examine it with respect to the above example.

Firstly, water management measures (scenarios, interventions) that are to be analyzed are formulated. In the example the measure is the intended groundwater extraction. But it may also be, for instance, the drainage of arable land, the felling of coniferous trees in order to reduce evapotranspiration, or the distribution of Rhine-water to new areas for the benefit of agriculture. Secondly, the hydrological effects of these measures may be calculated with hydrological models for the saturated zone (often a steady-state computation) and for surface waters. As these models compute water flows in the horizontal plane, I will refer to them as *spatial* hydrological models. Present hydrological models are capable of computing a spatial picture of hydrological variables, such as the phreatic groundwater level, the seepage intensity, and the discharge of rivers, rivulets and channels. For our problem we will use a steady-state (quasi-) three-dimensional groundwater model.

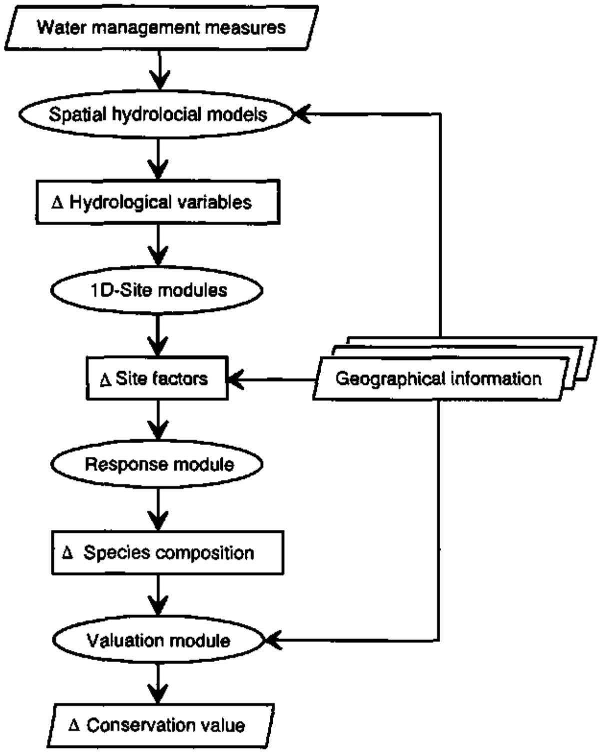

Figure 2.1: General process chain for an ecohydrological prediction. Input and output of data and variables are indicated by a rhomb or - in case of interim variables - by a box, models and modules by an ellipse, a direction of data flow by an arrow and a change in a variable by a triangle Δ.

The prediction for our nature reserve will probably be a fall in the groundwater level and a decrease in the intensity of upward seepage.

Though the chemical composition of the ground water is of ecological importance, models that are capable of computing this are scarce and have hardly been used in practice until now. The main reason is that these models require data - particularly data about the chemical characteristics of the subsoil - that are scarcely available (A. Leijnse, personal communication).

The next few steps in the prediction are carried out with so-called 'modules' (Fig. 2.1). I have called them this way because in some cases they are all attuned to one another during their development and they may be said to form a distinct whole which is, in fact, the core of the ecohydrological prediction.

Changes in the hydrological variables form the input to *site modules* which compute how site factors react on the hydrological changes. In contrast to the hydrological models for the saturated zone and the surface waters, the site modules do not contain spatial relationships: calculations are carried out for *plots*, i.e. one-dimensional and vertical representations of reality. Examples of site factors are: the depth of the groundwater table in spring, the soil moisture supply and the availability of nutrients. The predicted effects of the intended groundwater extraction on the site factors in the imaginary nature reserve would probably be (Fig. 1.9): (1) a lower availability of soil moisture, (2) a higher nutrient-richness and (3) a lower pH.

How the changes in these site factors, in their turn, influence the species composition of the vegetation, is predicted with a *response module*. This module consists of empirical or expert rules about the way species react on site factors. Animal organisms have not been incorporated in any ecohydrological response module yet, because the relationship between water management and animal organisms is too indirect and too complex. In the imaginary nature reserve, *Carex pulicaris* and *Cirsium dissectum* are predicted to disappear and new species, like common grasses, are assumed to take their place.

Sometimes the effect on the species composition is also evaluated in terms of nature conservation. In Fig. 2.1 this activity is indicated with the ellipse *valuation module*. Conservation valuation allows the outcomes for different species - or, if these are the classification units applied, for different vegetation types/ecosystem types - to be combined. This will yield results that are easy to interpret. If we would valuate on the basis of the criterion 'national rarity', there would be no doubt that the final output of the ecohydrological prediction for the presented nature reserve would be a conservation value that is lower than its current value.

Naturally, all models need geographical information, for instance about the transmissivity of the subsoil (spatial hydrological model), the soil type (site module) and about the vegetation (response module and valuation module).

The scheme of Fig. 2.1 will become more clear further on in this chapter. In Section 2.2 it will serve as a framework on the basis of which some approaches to ecohydrological modelling will be discussed. In Section 2.3 the general structure of DEMNAT will be described, as well as the modules and geographical information used for this model. Section 2.4 offers a discussion about DEMNAT's advantages and drawbacks. Additionally, possible improvements will be mentioned.

2.2 Discussion of Dutch ecohydrological models

2.2.1 Introduction

There are several models in the Netherlands that are referred to as 'ecohydrological models'. In a few publications, comparisons between these models have been made (Garritsen, 1993; Van der Veen & Garritsen, 1994; Van Wirdum, 1986; Wassen & Schot, 1992), but in my opinion each comparison has its drawbacks. An evaluation of these models is hardly possible for they differ too much in the kind of water management intervention, in the spatial scale, in the time span, as well as in the area they are meant for. Often, the emphasis of the comparison is on the *response module*, which constitutes but a part of the total prediction.

In this section I will discuss four models - each of which is somehow representative for a certain approach in Dutch ecohydrological modelling - and indicate how they fit into the general scheme of Fig. 2.1.

2.2.2 WAFLO

The first Dutch ecohydrological model was WAFLO (WAter-FLOra). This model is meant for the evaluation of the increase in groundwater extraction in the Pleistocene parts of the Netherlands (Fahner & Wiertz, 1987; Gremmen *et al.*, 1990; Reijnen & Wiertz, 1984). It comprises both the *response module* and the *valuation module* of Fig. 2.1.

For practical reasons, input variables have been used that can be obtained relatively easily by means of a hydrological model and with some rules of thumb from the Dutch $1 : 50,000$ soil map. The input to the response module consists for instance of the (present and future) average spring groundwater level (SGL) and the nitrogen mineralization. The response module is based on expert judgement and results in a list of species that will persist and a list of species that will - eventually - disappear. It contains 'if-then' rules applied to the indicator values of Ellenberg and Londo (Section 1.6), for example: "if the final SGL exceeds or equals 100 cm below soil surface, then species with an Ellenberg moisture indicator value of 6 or 7 will disappear".

A conservation value of the old and new species composition is obtained by weighing the species in proportion to their national rarity figures (which are derived from their UFK's) and, subsequently, adding them up (Reijnen *et al.*, 1981). In this way, the loss of conservation values is expressed in one figure, which facilitates a comprehensive mapping of results and a clear comparison of scenarios.

2.2.3 WSN

The WSN-armamentarium (Water-Site-Nature conservation value) (Kemmers, 1993; Van Beusekom *et al.*, 1990) stems from WAFLO. It is a set of allied models, devised for the assessment of adverse as well as favorable effects of water management measures on terrestrial fresh ecosystems. The core of the armamentarium is the NTM-model (Nature-Technical Model) (Gremmen, 1990), which comprises both the *response module* and the *valuation module* of Fig. 2.1.

Ellenberg's indicator scales for moisture, acidity and nutrient availability have all been reduced to three classes. The moisture-scale, for instance, to the classes 'wet' (indicator values $8 - 10$), 'moist' ($5 - 7$) and 'dry' ($1 - 4$). A combination of the classes results in a matrix of $3^3 = 27$ elements, each of which represents a certain site type, e.g.: 'wet, nutrient-rich, acid'. On the basis of Ellenberg's indicator values, ecological species groups are assigned to each site type. Furthermore, each site type is given a *potential conservation value*, calculated only once from the number of highly valued species (according to Arnolds, 1975) in the corresponding ecological group.

To facilitate predictions, the class boundaries have also been defined in physical terms. The boundary between 'wet' and 'moist' for instance, corresponds with a *SGL* of 20 cm below soil surface. Changes in site factors as a result of water management interventions are calculated by means of a number of site modules, for instance SWATRE (Belmans *et al.*, 1983) or WATBAL (Berghuys-Van Dijk, 1985) for the *SGL* and the soil moisture deficit in the growing season, ECONOM (Mankor & Kemmers, 1987) for the N-mineralization of the soil, and EPIDEM (Groenendijk & Van der Bolt, 1990) for the soil-pH. Computed changes in the site factors may bring about the crossing of class boundaries and, as a result, a new site type with its associated new potential conservation value.

2.2.4 ICHORS

A statistical approach is followed in ICHORS (Influence of Chemical and Hydrological factors On the Response of Species), which is a model for species of fresh surface waters and terrestrializing waters (Barendregt, 1993; Barendregt & Nieuwenhuis, 1993; Barendregt *et al.*, 1993). ICHORS comprises the *response module* of Fig. 2.1 only. In fact the model consists of a set of probability functions for species. Each of these functions describes the species occurrence probability in relation to abiotic variables, like surface water depth, Mg-content and *pH*. To illustrate this, Fig. 2.2 shows the occurrence probability of *Achillea ptarmica* (UK: Sneezewort; NL: Wilde bertram) as a function of both surface water level and chloride concentration. The probability functions are obtained as follows.

Data are collected in the study area only once: on many locations plants are recorded and abiotic variables are gathered, such as width and depth of the

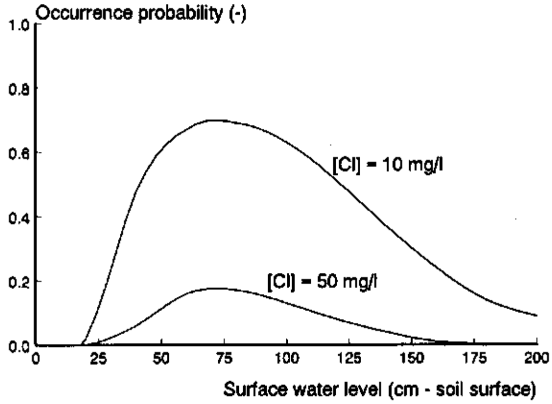

Figure 2.2: Occurrence probability of *Achillea ptarmica* as a function of surface water level at two different chloride concentrations (Barendregt & Nieuwenhuis, 1993).

surface water, pH, thickness of the sapropel layer and turbidity of the water. Also water samples are taken, which are submitted to a thorough chemical analysis (including for instance Si and Fe) in the laboratory. Then, with the computer program GENSTAT (Ter Braak, 1987) logistic multiple regression is carried out, fitting for each species a function trough the frequency occurrences in the data set. Over 25 variables are involved in the existing ICHORS-versions, but only those variables are selected for the function, that improve the 'goodness-of-fit'.

Because of the fact that so many variables are considered - irrespective of their supposed ecological importance - the model may be said to follow a *black-box approach*. This implies that for each region a special version of the model is made, with unique occurrence probability functions.

In scenario-analysis, ICHORS is applied by first estimating the new abiotic variables and, subsequently, computing the occurrence probability for each species by means of the probability functions.

2.2.5 MOVE

A model still in full development is MOVE (MOdel for the VEgetation). It is devised for the analysis of environmental stresses on a national scale (Latour & Reiling, 1993; Latour *et al.*, 1993). For the *response module* of MOVE, a method of Ter Braak & Gremmen (1987) is applied. This method combines the statistical approach of ICHORS with the indicator values of Ellenberg. Instead of abiotic field data, MOVE uses a large database of vegetation relevés to obtain information about the habitat of species. For each relevé, average Ellenberg indicator values for moisture regime, nutrient-availability, acidity and salinity are calculated. These averages are then processed with GENSTAT as if they were measured abiotic factors. The result is a set of equations, describing the occurrence probability of species as a function of Ellenberg's indicator values (Wiertz *et al.*, 1992).

Currently, the Ellenberg scales are calibrated to physical site factors, like soil-pH and N-mineralization (Alkemade *et al.*, 1996). These factors will have to be computed with dynamic site modules like the Simulation Model for Acidification Regional Trends, SMART (Kros *et al.*, 1995).

2.2.6 General discussion

Although the above overview had to be brief and incomplete, we now may draw a few conclusions. The first is that the response modules of the four models are based on the assumption that *vegetation is in equilibrium with its site*: with the predicted site conditions corresponds a clear-cut, new species composition. Succession from the old to the new equilibrium is not modelled, nor is the interaction between plants of different species incorporated in the prediction. Such simplifications of reality are necessary for practical applications in policy analysis. They arise from a lack of both ecological knowledge and data. Dynamic models with interactions between plants do exist for relatively simple ecosystems like moist heathlands (Berendse, 1988), but for policy analyses their practical use is limited.

A second conclusion is that for all four models it is assumed that *the site of a vegetation is homogeneous*, i.e. that all the plants have their roots in the same environment. This may be true for many ecosystems but - as I mentioned in Section 1.6.4 - it is definitely not the case for e.g. quagfens and some hayfields on wet, nutrient-poor, weakly acid soils. Also this simplification of reality is made in order to get a model that is usable in practice.

A third conclusion - already mentioned in the beginning of this section - is that all models cover *different fields of applications*: WAFLO, for instance, is meant for the evaluation of negative effects of water management measures (in the Pleistocene areas of the Netherlands), whereas ICHORS is intended to be used both for negative and favorable effects (in small surface waters).

Finally, we may conclude that the models *differ considerably as to their practical applicability*. WAFLO is clearly constructed to be used in practice, and indeed this model has been applied in several cases for the evaluation of a proposed expansion of groundwater extraction (e.g.: De Jong & Van der Most, 1985; Fahner, 1993). The applicability of WSN is limited because some of its modules are only a first rough approach whereas others require relatively many input data. It is significant that, until now, WSN has only been used for the compilation of sample cards of a very limited number of ecosystems (10) (Kemmers, 1990; Van Beusekom *et al.*, 1990). Moreover, for these sample cards the effect on the conservation value was judged by experts (Hochstenbach & Gremmen, 1989), instead of by the NTM-model. However, my remarks on WSN are not meant to criticize the armamentarium. To my mind, the development of WSN was especially of scientific importance, that is to say: a first step towards a more complicated and process-orientated model. In fact, some of the results established in the building of WSN, have been used for the making of DEMNAT. Currently, components of the WSN-armamentarium are upgraded in order to serve a model of the catchment area of the Drentse Aa, a rivulet in the province of Drenthe (Kemmers, 1994; Prins, 1995; Prins *et al.*, 1996). However, in my opinion the utility of ICHORS is very limited because this model requires input data that are difficult, if not impossible, to obtain (like the future concentrations of HCO_3, Si and Fe). Moreover, ICHORS does not provide a procedure to aggregate the results (occurrence probabilities of species) into for instance a conservation value.

What I criticize, is the statistical approach of ICHORS. As the data of ICHORS are not collected randomly, the computed occurrence probability has no meaning in reality. Also the ecological amplitude, the optimum and the shape of the probability function may be affected by the sampling method. When for instance a certain species is oversampled at the low values of a variable, the probability function will tend to a log-normal shape. Another point of criticism is the black-box approach, which may lead to apparent correlations and, in this way, to unusable probability functions. Apparent correlations may be avoided by exclusively considering those site factors that are most important for the species composition of the vegetation. In this respect I prefer MOVE which, like WAFLO and NTM, solely uses such important factors. But the probability functions of MOVE are based on data that are by no means collected randomly. Moreover, as Runhaar *et al.* (1994) already pointed out, it is not allowed to use average Ellenberg-indicator values of non-randomly collected vegetation relevés: in this case, averaging may lead to an artificial and undesirable shift in the computed optimum of species.

2.3 DEMNAT: a national model for ecosystems

2.3.1 History

DEMNAT is a national prediction model, meant for analyzing the effects of water management on ecosystems. The first version - DEMNAT-1 (Witte, 1990; Witte & Claessen, 1991; Witte *et al.*, 1989; Witte *et al.*, 1992) - was used in scenario analyses (Claessen, 1990; Claessen & Witte, 1991; Claessen *et al.*, 1991) for the benefit of the 'Third National Policy Document on Water Management' (Ministerie van Verkeer en Waterstaat, 1989), which outlines the Dutch Government's future water management policy.

An improved second version - DEMNAT-2.0 - was issued in 1992 (Witte *et al.*, 1993; Witte *et al.*, 1994). DEMNAT-2.0 is the product of close cooperation between scientists of several research institutes: the Centre of Environmental Sciences (CML) and the National State Herbarium both of Leiden University; the Department of Water Resources of Wageningen Agricultural University; the Institute for Inland Water Management and Waste Water Treatment (RIZA) of the Ministry of Transport and Public Works; and, finally, the National Institute of Public Health and Environmental Protection (RIVM) of the Ministry of Housing, Physical Planning and Environmental Protection. DEMNAT-2.0 has been applied in several analyses, such as in the impact assessment for the 'National Policy Plan on Drinking Water and Industrial Water Supply' (Beugelink *et al.*, 1992; Claessen & Beugelink, 1995; Claessen *et al.*, 1996; Ministerie van Volkshuisvesting, Ruimtelijke Ordening en Milieubeheer, 1995), and for the analysis of measures against the desiccation of the Netherlands (Beugelink & Claessen, 1995; Beugelink & Claessen, 1996).

Since 1992 the model has been subjected to several modifications and improvements. Regional versions became available, which were used to analyze the ecological effects of groundwater extraction (Hoogeveen, 1994; Poelman & Pakes, 1995; Poelman & Van Ek, 1996a; Poelman & Van Ek, 1996b). But also the national version was upgraded to a DEMNAT-2.1 (Van Ek *et al.*, 1996; Van Ek *et al.*, 1998). This version was used to analyze scenarios (Kors *et al.*, 1997; Ministerie van Verkeer en Waterstaat, 1997a) for the benefit of the 'Fourth National Policy Document on Water Management' (Ministerie van Verkeer en Waterstaat, 1997b).

2.3.2 General approach

The makers of DEMNAT aimed at a practically applicable model, rather than at a scientific model, producing results which have to be interpreted as the 'best possible judgement', given the current knowledge and availability of data. Referring to Fig. 2.1, DEMNAT consists of a *site module*, a *response module* and a *valuation module*. Moreover, it contains a *geographical schematization of the Netherlands* with e.g. information about the distribution of ecotope groups in the Netherlands. *Ecotope groups* are the ecosystem types of DEMNAT. Each ecotope group consists of one or more ecotope types according to the ecotope system, described in Section 1.6. Table 4.2 (Section 4.2.1) lists the ecotope groups that are involved in DEMNAT. An *ecotope map* is a map that shows the *completeness* ('botanical quality', Section 1.3) of a certain ecotope group per km-square (color Figs. 4.26-4.53).

DEMNAT's predictions are based on the assumption that vegetation is the biotic ecosystem variable, affected by water management in the most direct way. The predictions relate to the medium long term (about 20 years), which is considered to be a relevant time span in policy making. The basic spatial computation unit of DEMNAT is the *ecoplot*, generated by combining various maps, e.g. on ecotope groups, soil type, and hydrology.

Fig. 2.3 - which was made using the general scheme of Fig. 2.1 as a guide - gives the process chain for a computation with DEMNAT. The box 'geographical information' of Fig. 2.1 has been omitted because Fig. 2.3 concerns only one ecoplot. Moreover - in contrast to the simplified general structure of Fig. 2.1 - Fig. 2.3 reveals a feedback (from 'ΔSGL' to 'Translation rules'). Finally, the ellipse '1-D site module' has been split up into the ellipses 'MOZART' and 'Translation rules'.

The prediction chain is as follows. Water management measures are implemented in the NAtional GROundwater Model NAGROM, which computes a resulting seepage intensity s, that is: the vertical groundwater flow from the first (regional) aquifer to the phreatic groundwater. This seepage intensity is used as a lower boundary condition in MOZART, a model for the unsaturated zone. The output of MOZART for DEMNAT is: (1) a change in the average Spring Groundwater Level (ΔSGL), (2) a change in the average Spring Surface water Level of ditches (ΔSSL), (3) a change in the intensity of upward seepage (Δs). Of course these changes are computed in two runs: one with the intended measures and one without them. A change in the average spring groundwater level or in the seepage intensity influences the availability of nutrients and the acidity of the soil. These effects are estimated with what is indicated in Fig. 2.3 as 'Translation rules', i.e. rules that are based on empirical field data, process studies described in literature and expert judgement. The effects are expressed as a change in N-mineralization (ΔN) and a change in soil-pH (ΔpH) respectively.

The supply of surface water from rivers and channels influences the salinity and the nutrient-availability of local surface waters. In this way, it may affect water

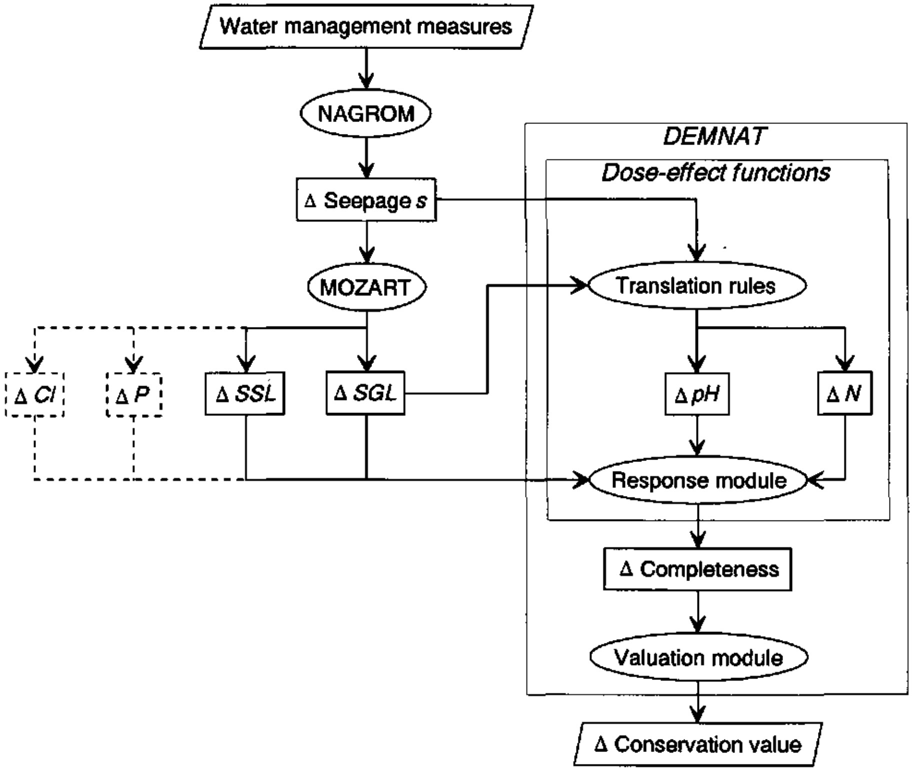

Figure 2.3: General process chain for a prediction with DEMNAT-2.1, concerning the basic computation unit of DEMNAT, the ecoplot. Dotted lines indicate the parts that are under construction.

vegetations and terrestrializing vegetations. In the near future this influence will also be computed by MOZART by means of a distribution module for surface water flow. The outcomes will be: (1) a change in chloride content (ΔCl) and (2) a change in phosphorus content (ΔP).

Subsequently, how ΔSSL, ΔSGL, ΔpH, ΔN, ΔCl and ΔP affect the completeness of the ecotope group is computed with a response module. Finally, a valuation module may be used in order to express the effect in a conservation value figure.

The previous description gives an idealized and simplified picture of DEMNAT. In reality, it is a complex model, which is mainly caused by a shortage of suitable geographical data and empirical ecological data. Various solutions have been proposed to make optimal use of these data, some of which will be presented in this publication (Chapter 4 and 5), while other solutions have been discussed extensively elsewhere (Bleij & Witte, 1996; Klijn *et al.*, 1996; Klijn *et al.*, 1997; Runhaar *et al.*, 1996b; Runhaar *et al.*, 1997c; Runhaar *et al.*, 1997b; Van Ek *et al.*, 1996; Van Ek *et al.*, 1998; Witte *et al.*, 1994).

In my description of DEMNAT, the emphasis will be on the ecological effects of a change in spring groundwater level, ΔSGL. Firstly, because these effects are modelled in a satisfactory way; secondly because this dose has the greatest impact on the vegetation (Arts *et al.*, 1997).

2.3.3 Hydrological input

The hydrological input for DEMNAT may be generated by the models NAGROM and MOZART. These are combined by means of a special GIS interface procedure - MONA (MOzart–NAgrom) - which accounts for spatial scaling problems (Fig. 2.4). NAGROM and MOZART will first be described, upon which the interaction between these models will be commented on by means of MONA.

NAGROM

NAGROM is a *national steady-state model* for the *saturated zone* (De Lange, 1991a; De Lange, 1991b; De Lange, 1996), based on the Analytic Element Method (Strack, 1989). Within this method, the groundwater system is simulated by analytical elements, each element representing a feature in the groundwater system, for instance a resistance layer, a crack or a well. Each element is represented by an analytical solution of the differential equation for two-dimensional groundwater flow. As this differential equation is linear, it allows the application of the principle of superposition, which implies that the solutions of various elements may be added or subtracted freely.

Within NAGROM, the Netherlands' groundwater system has been schematized into several aquifers and aquitards. In the aquifers, groundwater is supposed to flow horizontally (Dupuit-Forcheimer assumption), whereas in the aquitards only vertical flow is assumed.

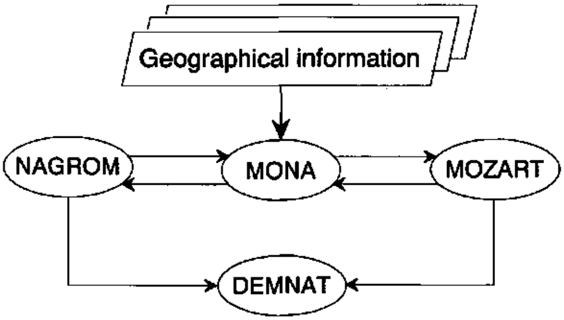

Figure 2.4: The groundwater model NAGROM and the unsaturated zone model MOZART are coupled by the GIS-interface MONA. When NAGROM and MOZART are atuned to each other by MONA, their output is used by DEMNAT.

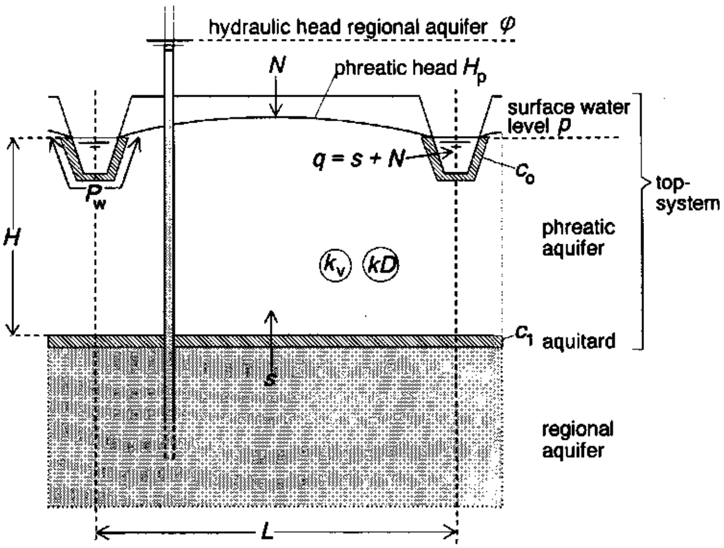

Figure 2.5: Conceptual model of the topsystem within NAGROM. Explanation: see text.

In a flat, deltaic lowland like the Netherlands, the flow of phreatic groundwater is largely determined by small-scale characteristics of the drainage system, such as the depth of the surface water level and the distance between the ditches. As it is both impossible and superfluous to model each ditch and each drain pipe separately in NAGROM, an approach has been developed in which the separate drainage features are put together. The conceptual model for this approach is given in Fig. 2.5.

The upper boundary consists of a topsystem which is placed on a regional aquifer. This topsystem in its turn, consists of a phreatic aquifer - with equidistant and parallel surface waters (the drainage system) - which is placed on an aquitard. A Cauchy upper boundary condition is used to describe the relation between the regional aquifer and the drainage system (De Lange, 1996; De Lange, 1997):

$$s = \frac{\phi - p}{c^*} - \frac{c_{\mathrm{dran}}}{c^*} N \tag{2.1}$$

where:

s = seepage from the regional aquifer $[\mathrm{L\,T^{-1}}]$
ϕ = head in the regional aquifer [L]
p = surface water level [L]
c^* = feeding resistance [T]
c_{dran} = drainage resistance [T]
N = groundwater recharge $[\mathrm{L\,T^{-1}}]$

In this equation, both seepage s from the regional aquifer through the aquitard and the hydraulic head ϕ in the regional aquifer are unknown variables that have to be solved by NAGROM, whereas surface water level p, feeding resistance c^*, drainage resistance c_{dran}, and groundwater recharge N are specified.

The *feeding resistance* c^* is a total resistance against vertical and horizontal flow of groundwater from the regional aquifer to the drainage system. It is the sum of a series of five resistances [T]:

1. A vertical resistance of the aquitard c_1.
2. A vertical resistance of the phreatic aquifer, which is equal to $\frac{H}{k_v}$, where: H = distance [L] between aquitard and surface water level p; k_v = vertical hydraulic conductivity of the phreatic aquifer [L T^{-1}].
3. A horizontal resistance between the phreatic groundwater and the surface waters. This resistance depends on the distance between the surface waters L [L] and the transmissivity of the phreatic aquifer kD [L^{2}T^{-1}].
4. A radial resistance near the surface waters c_{rad}.
5. A resistance of the bottom and sides of the surface waters c_o.

The sum of the first two resistances refelects the vertical resistance c_1' [T] of the top sytem below p:

$$c_1' = c_1 + \frac{H}{k_v} \tag{2.2}$$

The *drainage resistance*[1] c_{dran} is the sum of the last three resistances. It is used to describe the relation between the average phreatic head $\overline{H}_p$ [L] and the groundwater flow to the surface waters - i.e. the *drainage* q [L T^{-1}] - according to: $q = \frac{\overline{H}_p - p}{c_{\mathrm{dran}}}$. Of course, q equals $s + N$.

De Lange (1996) combined all five resistances into one expression for the feeding resistance:

$$c^* = c_o \frac{L}{P_w} + c_{\mathrm{rad}} L + \left(c_1' + c_o\right) \frac{L}{2\lambda} \coth\left(\frac{L}{2\lambda}\right) \tag{2.3}$$

where (Fig. 2.5):

P_w = wetted perimeter [L]
λ = leakage factor [L]: $\lambda = \sqrt{kD\, c_1'}$

And he derived c_{dran} from:

$$c^* = c_1' + c_{\mathrm{dran}} \tag{2.4}$$

---

[1] De Lange defines c_{dran} as the resistance between the *average* phreatic head $\overline{H}_p$ and the surface water level p. In doing so, he deviates from the common definition of Ernst (1956), who relates the drainage resistance to the head *right in between* the ditches (Van Drecht, 1997).

Eq. (2.1) is applicable for all types of areas within the Netherlands. In areas with abundant surface waters, such as polders, c_1' is often much higher than c_{drain}, so that $c^* \approx c_1'$ and $c^* \gg c_{\mathrm{drain}}$. As a consequence, the second right-hand term in Eq. (2.1) is negligibly small with respect to the first one, and the equation may be simplified to: $s \approx \frac{\varphi - p}{c_1^*}$.

Conversely, in infiltration areas surface waters are as good as absent. As L is large in such areas, c_{drain} is large too and c_1' becomes negligible compared to c^* so that $c^* \approx c_{\mathrm{drain}}$ and the first right-hand term in Eq. (2.1) becomes negligible. As a result, Eq. (2.1) diminishes to: $s \approx -N$.

In for instance brook valleys, an intermediate situation exists, in which both terms are important (Hoogeveen & Vermulst, 1997).

Fig. 2.5 only shows one drainage system, but in reality two drainage systems with a given surface water level p and one or two corresponding drainage resistances c_{drain} are distinguished in NAGROM. The primary drainage system represents groundwater flow to rivulets and large ditches, the secondary system deals with the flow to small surface waters (ditches and trenches) and drain pipes. If the phreatic surface exceeds the depth of the secondary drainage system, both drainage systems are taken into account. In all other cases, only the primary drainage system functions.

Within NAGROM, the area is covered with quadrilateral elements of approximately 1 to 5 km^2. An element is considered uniform: p, c^*, c_{drain} as well as N are assumed to be constant within an element. Therefore, the lay-out of the elements clearly reflects the different parts of the hydrological system, such as valleys and infiltration areas.

For each element and for each aquifer, NAGROM computes the hydraulic head as an analytical *function* of the position in the horizontal plane. So NAGROM is capable of producing hydraulic heads *for any desired location.*

MOZART

MOZART (Arnold, 1995; Bos *et al.*, 1997) is a *one-dimensional* model, which simulates *vertical transport* of water in the *unsaturated zone.* For applications on a national scale, a *pseudo-steady-state* approach has been developed, solving Richard's equation with a time-step on decade basis (Abrahamse *et al.*, 1982).

MOZART performs computations for *homogeneous* hydrological units, of which the parameters are based on a unique combination of meteorology, vegetation, soil type, seepage intensity and drainage situation. For nation-wide applications these *hydroplots* consist of 500 m grid-cells, which are the result of an overlay of various geographical databases.

A *hydroplot* is a one-dimensional system, consisting of an effective root zone and a subsoil (Fig. 2.6). The upper boundary condition is given by the precipitation rate and the potential evapotranspiration rate. The lower boundary condition by

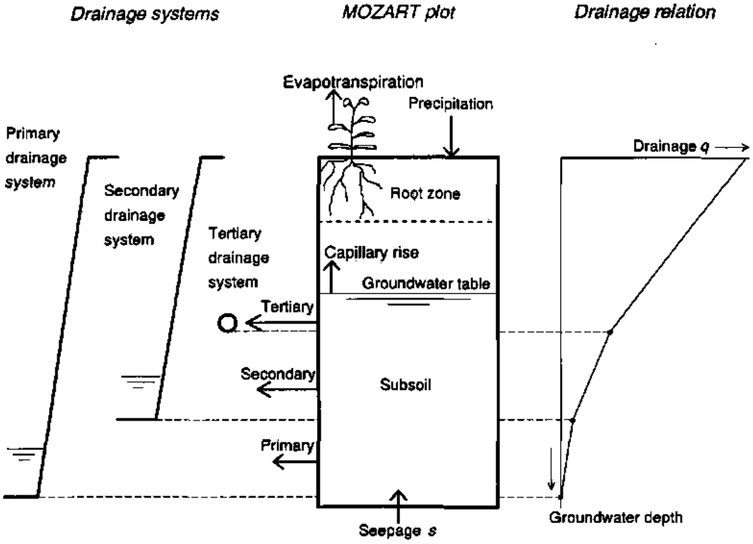

Figure 2.6: National schematization of the subsoil and the drainage relation used in MOZART. After: Vermulst *et al.* (1996).

the seepage intensity s $[\mathrm{L\,T^{-1}}]$ between the regional aquifer and the hydroplot, which is output of MONA. The flow of groundwater to the surrounding surface waters and drain pipes - the drainage - is described by means of a relation of three linear segments (Fig. 2.6). The slopes of these segments reflect the drainage resistances c_{dran}. The intersections of the segments reflect the drainage bottoms of respectively the primary system (canals and rivulets), the secondary system (ditches) and the tertiary system (trenches and drain pipes).

For DEMNAT, MOZART computes per 500 m square an average spring groundwater level and an average spring surface water level (SGL and SSL) on the basis of meteorological data of 1985, which may be considered as an average year. Vermulst and Hoogeveen (personal communication) estimate that the average absolute difference between computed SGL and actual SGL ranges from about 0.2 m in flat areas, such as polders, up to more than 1 m in elevated hilly areas and areas with boulder clay (province of Drenthe). This estimation is based on a comparison of computed phreatic heads with the groundwater table classes on the $1:50,000$ Dutch soil map.

MONA

MOZART is a more detailed approach of the top system already modelled by NAGROM. The GIS-interface MONA is based on similarities in the schematizations of both models. Consequently, for both models the groundwater recharge N, the drainage resistances c_{dran} and the surface water levels p have to be identical *on average.* In addition, for both models, the seepage s from the first, regional aquifer to the topsystem is given by Eq. (2.1).

Values of p, c^*, N as well as c_{dran} are derived from various GIS databases and subsequently transformed by MONA to average values for each NAGROM element covering the top system. To this end, averaging procedures have been developed that account for spatial scaling problems (Vermulst *et al.*, 1996). The same scaling procedures are also used for the benefit of MOZART, to generate average values per 500 m square of p, N and c_{dran}. Moreover, as a lower boundary condition MOZART requires a steady-state seepage flux s per 500 m square. This flux is calculated on the basis of Eq. (2.1), using heads ϕ per 500 m square that are computed with NAGROM.

MOZART computations may result in N-values that, when averaged per NAGROM-element, deviate too much from the value, originally used in NAGROM. In that case, the N-values of NAGROM are adjusted and new NAGROM and MOZART computations are carried out. Carrying out the MONA procedure once is already sufficient in most cases. One repetition of the procedure leads to satisfactory results in nearly all remaining cases (Vermulst *et al.*, 1996).

The color map of Fig. 2.11 gives a nation-wide picture of the computed seepage flux on a 500 m square grid (present situation).

Other hydrological models

Strictly speaking, DEMNAT may get its hydrological input from any hydrological model and any spatial computation unit desired. For regional applications, the resolution of the hydroplots of MOZART will be refined to 100 × 100 m (Vermulst *et al.*, 1996). As most of the databases which MOZART uses for its schematization are on a $1:50,000$ scale, this refinement can relatively simply be implemented in the MONA procedure (Vermulst *et al.*, 1996). For applications of MOZART on a local scale, the model uses a *dynamic* approach, solving Richard's equation with a numerical scheme (Borsboom & Prinsen, 1995). Moreover, in this local version the subsoil may be schematized into several layers, each of which has its own soil physical characteristics. Perhaps in the near future, MOZART will be replaced by SWAP (Soil-Water-Atmosphere-Plant) (Van Dam *et al.*, 1997), a more sophisticated dynamic model for the unsaturated zone which stems from SWATRE (Belmans *et al.*, 1983; Feddes *et al.*, 1978).

It is also possible to use polygonal hydroplots, instead of squares. In several regional studies, hydroplots were obtained by making an overlay of polygonal maps of soil, groundwater table depth and land use (Hoogeveen, 1994; Van der Voet & Witte, 1991; Vermulst, 1992). This was also done in a study of the ecological effects of groundwater extraction in the eastern part of the province of Gelderland (Poelman & Pakes, 1995; Poelman & Van Ek, 1996a; Poelman & Van Ek, 1996b). In this study, the hydrological input for DEMNAT was computed with the SIMGRO model (SIMulation of GROundwater flow and surface water levels) (Querner, 1993).

As for DEMNAT-applications on a national scale, the combination of NAGROM, MONA and MOZART may be substituted by LGM (UK: National Groundwater Model; NL: Landelijk Grondwater Model) (Pastoors, 1992; Pastoors *et al.*, 1993). LGM generates output for each combination of km-square and groundwater depth class GWT (Table 2.1). LGM was used in combination with DEMNAT in a national impact assessment of groundwater extraction (Beugelink *et al.*, 1992; Claessen & Beugelink, 1995; Claessen *et al.*, 1996).

2.3.4 Geographical schematization of the Netherlands

DEMNAT uses various geographic maps with hydrologically and ecologically relevant information. One of them contains the contours of the hydroplots. When DEMNAT is connected with NAGROM/MOZART, these hydroplots measure 500 × 500 m. Other maps will be discussed below.

Information on ecosystem types: ecotope groups

Chapter 4 of this publication describes how distribution maps of 28 ecotope groups have been derived from FLORBASE-2c, a database containing km-square data on the presence of indigenous plant species in the period 1975 – 1995 (Section 1.4.2). On such a map each km-square is classified into one of the following completeness classes: 'noise/insufficient data', 'low', 'high', 'very high'. This classification is based on the occurrence in FLORBASE-2c of records of plant species indicative for the ecotope group in question. The classes are depicted on the ecotope maps by the colors grey, blue, yellow and red respectively (see the color maps of Fig. 4.26-4.53). Only well-investigated km-squares are used in the geographical schematization (Fig. 4.14, Section 4.4.2).

Table 4.2 (Section 4.2.1) describes the 28 ecotope groups. It also contains information about the ecotope types each ecotope group is derived from, as well as about the corresponding color map. Eighteen out of the 28 groups are taken into account in DEMNAT, since they are susceptible to changes in water management. In Table 4.2 these ecotope groups are marked with a 'd'.

For computation ends, it is not practical to use the ordinal completeness classes. Therefore, the completeness of each ecotope group in a km-square is expressed as a fraction C (Section 4.4.3).

Information on soil and groundwater: ecoseries types

DEMNAT needs information on soil characteristics (e.g. soil-pH and soil texture) because the reaction of a vegetation on hydrological changes is largely dependent on controlling soil factors. This may be illustrated by for instance a meadow vegetation, growing on a wet, nutrient-poor and weakly acid site (site type X22 of the ecotope system). Such a vegetation may be found on certain sandy soils with an organic toplayer, but also on sandy soils without an organic toplayer. Because of the extra release of N due to the mineralization of organic matter, a

Table 2.1: The six Ground Water Table classes (*GWT*'s) used in DEMNAT, with an indication of the ecologically relevant groundwater table depth in spring and the original Groundwater table classes (*Gt*'s) on the 1 : 50,000 soil map of the Netherlands (De Waal, 1992).

GWT	Description	Groundwater table depth in spring (cm below surface)	*Gt*
0	Surface water	Not relevant	0
1	Very shallow	< 30	I
2	Shallow	< 55	II
3	Moderately deep	35-65	II*, III, III*, V, V*
4	Deep	55-100	IV, VI
5	Very deep	> 100	VII, VII*

fall in the groundwater table is bound to be more harmful to the vegetation on soils with an organic toplayer.

In DEMNAT, a simplified soil classification is applied. Soils that react in a similar way on hydrological changes - like a lowering of the groundwater table - and that are comparable as to ecological relevant conditioning factors - such as soil texture and $CaCO_3$ content - have been merged into *ecological soil types*. For DEMNAT, the 2000 units of the 1 : 50,000 Soil Map of the Netherlands (Soil Survey Institute, Wageningen, in about 60 volumes) have been merged into 48 ecological soil types (Klijn *et al.*, 1996; Klijn *et al.*, 1997). These soil types have been mapped nation-wide, with an accuracy of 0.0025 km^2.

Information about ecotope groups is available with a resolution of km-squares. Because of the fact that in most km-squares several soil types occur, we have to determine on which soil type or on which soil types each ecotope group is most likely to occur. To *down-scale* the information about ecotope groups, i.e. to locate each ecotope group within its km-square, not only the maps with ecological soil types are used, but also information about the depth of the groundwater table and about seepage, as will be described below.

The 1 : 50,000 soil map of the Netherlands contains information about the depth and the dynamics of the groundwater table. This information is presented as Groundwater table classes (*Gt*'s) on a 12-points scale of Roman numbers and asterisks, based on the averagely highest and the averagely lowest groundwater levels in cm below soil surface. The *Gt*-scale - originally devised for agricultural use - was reduced to 5 ecologically relevant Ground Water Table classes (*GWT*'s) (De Waal, 1992), see Table 2.1. DEMNAT uses the nation-wide *GWT*-map that, like the soil map, has an accuracy of 0.0025 km^2. The color map of Fig. 2.12 shows the national distribution of GWT2 per km-square.

Upward seepage was mapped by Klijn (1989) on the basis of various sources, for instance maps with hydraulic heads and stream lines in aquifers. In DEMNAT, the following four ecologically relevant seepage classes are distinguished:

(1) no exfiltration, (2) exfiltration of lithotrophic groundwater, (3) exfiltration of brackish groundwater, (4) exfiltration of saline groundwater. For each km-square a rough estimation of the surface areas of the seepage classes is given. These areas have been ascribed to one or to several ecological soil types on the basis of assumed general relationships between seepage and soil type (Klijn *et al.*, 1997).

The ecological soil types, the GWT's and the seepage classes were joined together by Klijn *et al.* to form ecoseries types (Klijn *et al.*, 1996; Klijn *et al.*, 1997). 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: soils of similar or comparable parent material within a climatically homogeneous area, resulting in similar site conditions during a considerable time span (Klijn *et al.*, 1996). Theoretically, the number of $48 \times 4 \times 6 = 1152$ ecoseries types can be distinguished, but in practice - when the maps on soil, groundwater table depth and seepage are joined together - no more than 435 relevant combinations are found.

For each ecoseries type the supposed frequency distribution of operational site types is given by Klijn *et al.* in the form of a *site diagram* (Klijn *et al.*, 1996; Klijn *et al.*, 1997). Table 2.2 shows the general scheme of a site-diagram. This scheme has the same format as Table 1.3. In Table 2.3 some examples of ecoseries types are given, namely:

- Primary oligothrophic peat (ecological soil type V01) with both a shallow and a very shallow groundwater table (resp. GWT2 and GWT1) and no upward seepage (seepage class 1). Especially ombrotrophic bogs found in nature reserves only.
- Calcareous sands without a topsoil (Z17) and with both a deep and a very deep groundwater table (GWT4 and GWT5) and no upward seepage (class 1). Especially the coastal dunes of the south-western part of the Netherlands, with a very low availability of soil moisture, which is reflected by a xerophytic vegetation of nutrient-poor conditions.

From Table 2.3 it appears that on oligotrophic peat with high groundwater levels, wet, nutrient-poor and acid conditions prevail, whereas on calcareous sands with a low groundwater level, neutral and basic conditions may be expected.

By multiplying the estimated shares of each site type according to the site-diagrams with data concerning the surface area of ecoseries in each km-square, it is possible to calculate the surface area of site types in each km-square. To this end, Klijn *et al.* used means for the various frequency classes in the site-diagrams and afterwards normalized the results to 1 km^2 (Klijn *et al.*, 1996; Klijn *et al.*, 1997). In this way, they obtained per km-square an estimation of the potential surface area of site types. The color maps of Fig. 2.13 and 2.14 show the result in the form of two site maps: one of a common site type (X27) characteristic of (lowland) peats and brook valleys, and another of a rare site type (X63), characteristic of the coastal dunes and fluvial sands (of levees).

The site-diagrams are based on expert judgement by a team of five vegetation ecologists and soil scientists who are familiar with both the ecotope system and

Table 2.2: Site-diagram (general scheme), derived by generalizing ecotope types according to similar abiotic site conditions (Klijn *et al.*, 1996). The absence of a code means that the site type does not occur in the Netherlands.

	Fresh					Brackish	Saline
	Nutrient-poor			Mod. nutrient-rich	Very nutrient-rich		
	Acid	Neutral	Alkaline				
Water	X11	X12	X13	X17	X18	bX10	zX10
Wet	X21	X22	X23	X27	X28	bX20	zX20
Moist	X41	X42	X43	X47	X48	bX40	
Dry	X61	X62	X63	X67	X68	bX60	

Table 2.3: Examples of site diagrams of four ecoseries (all of seepage class 1: no upward seepage), showing the expected frequency of site types in classes* (Klijn *et al.*, 1996). The format of the site diagrams is given in Table 2.2.

Ecological soil type 'Primary oligotrophic peat, not mineralized' (V01) with very shallow (GWT1, left) and shallow (GWT2, right) groundwater table

3	+	−	+	+	−	−
4	+	−	+	+	−	−
−	−	−	−	−	−	−
−	−	−	−	−	−	−

1	+	−	+	+	−	−
4	+	−	+	+	−	−
2	+	−	+	+	−	−
−	−	−	−	−	−	−

Ecological soil type 'Calcareous sands without topsoil' (Z17) with deep (GWT4, left) and very deep (GWT5, right) groundwater table

−	−	−	−	−	−	−
−	+	+	+	−	−	−
−	2	1	1	−	+	−
−	3	3	1	−	+	−

−	−	−	−	−	−	−
−	+	+	+	−	−	−
−	+	+	+	−	+	−
−	3	3	1	−	+	−

*Frequency classes:	−	Very unlikely	2	0.05-0.24	4	0.50-0.74
	+	Likely/ < 0.01	3	0.25-0.49	5	0.75-1.00
	1	0.01-0.04				

the Dutch soil-map, and who also have adequate knowledge about field circumstances in large parts of the country. For each ecoseries type, consensus was required on a semi-quantitative estimation of the abundance of all operational site types. This estimation - expressed in frequency classes (Table 2.3) - is valid for the present GroundWater Table class GWT and for undisturbed conditions only. This may be understood as already having been influenced by the intensive drainage in the present century.

Merging of geographical information: the ecoplot

Summarizing, we have: (1) a nation-wide map of *hydroplots* of 500 m squares, (2) nation-wide maps of *ecotope groups* derived from floristic information with a resolution of 1 km^2, and (3) a nation-wide map of *ecoseries types*, derived from abiotic information and accurate to 0.0025 km^2. To achieve homogeneous calculation units, an overlay was made of these maps (Nienhuis, 1994; Van Ek *et al.*, 1998; Witte *et al.*, 1994). These units - *ecoplots* - are stored in a file which is read sequentially when making a computation. DEMNAT will give its results per ecoplot. These results can be merged to results per ecotope group, or, for instance, per km-square. The use of ecoplots provides readily accessible information and has the advantage of a short computation time (about 15 minutes for a scenario on a 90 MHz Pentium PC).

By way of example Table 2.4 shows the description of a certain ecoplot in the ecoplot file. Each ecoplot (one record) is described by codes for the hydroplot (first column), the ecotope group (second column), and for the ecoseries type (third column). The coordinates of the hydroplot (NAGROM/MOZART: x- and y-coordinates of the lower left corner of the 500 m square in hm) give access to the hydrological input. Information about the kind of ecotope group and ecoseries is necessary to compute how the completeness fraction C (fourth column) changes as a result of the hydrological input (Section 2.3.5). Furthermore, the description contains information about the following.

The *fraction of the ecotope group* that is ascribed to the ecoplot (fifth column). The ecoplot results have to be multiplied with this fraction in order to count each ecotope group just once, in spite of its presence within the km-square in several ecoplots. To calculate this fraction, it is assumed that each ecotope group is distributed over the ecoplots proportionate to the potential surface areas of its associated site type. Let us consider for instance a km-square consisting of three ecoplots of 10, 40 and 50 ha. If we assume that - according to the site-diagrams - the frequency classes (Table 2.3) of site type X21 are 4 ($0.50 - 0.75$), 1 ($0.01 - 0.05$) and 2 ($0.05 - 0.25$) respectively, then, with average frequencies of 0.625, 0.03 and 0.15, the potential areas of X21 in these ecoplots will be 6.25, 1.2 and 7.5 ha respectively (total: 14.95 ha) and the fraction of K21 ascribed to each ecoplot is 0.42, 0.08 and 0.50 respectively.

The *occurrence of perched water tables* (last column).
In some places in the Netherlands, a vegetation typical of wet/moist circumstances may occur on soils with a very deep groundwater table. One explanation

Table 2.4: Example of an ecoplot description. Explanation: see text.

Hydroplot (coor. in hm)	Ecot. group	Ecoseries (Soil-GWT-Seep.)	Completeness C	Fraction ecotope group	Perched water table
2055 5650	K21	V01-2-1	0.85	0.42	0

of this phenomenon is a perched water table (e.g. in some podzol soils in the Pleistocene parts of the Netherlands). Another explanation is that it concerns a soil, where the availability of water in the root zone is high (loess soils in the southern part of the province of Limburg). As these locations cannot be influenced by water management, ecoplots with a perched water table (or much available soil water) are marked with a switch (0 = no perched water table, 1 = perched water table), and, consequently, no effect on the ecotope group is computed for them.

2.3.5 Effects of hydrological changes on ecotope groups

Introduction

In Section 2.3.3 I described how hydrological models generate the input Δs, ΔSGL and ΔSSL for DEMNAT. In the near future, also changes in the chloride content (ΔCl) and in the phosphorus content (ΔP) of local surface waters will be modelled. The effect of these *doses* on the completeness fraction C is computed in two steps, see Fig. 2.7 (Runhaar *et al.*, 1996b; Runhaar *et al.*, 1997c).

Firstly, it is determined which changes in operational site factors are expected to occur, given a certain dose. These changes depend on ecological soil characteristics, which can be deduced from the ecological soil type. Secondly, empirical relationships between ecological species groups and operational site factors are used to predict how these changes will affect C. The current DEMNAT-version makes use of *dose-effect functions* in which both steps are integrated. These functions were computed for each possible combination of ecological soil unit and ecotope group (Runhaar *et al.*, 1996b; Runhaar *et al.*, 1997c). By way of example Fig. 2.8

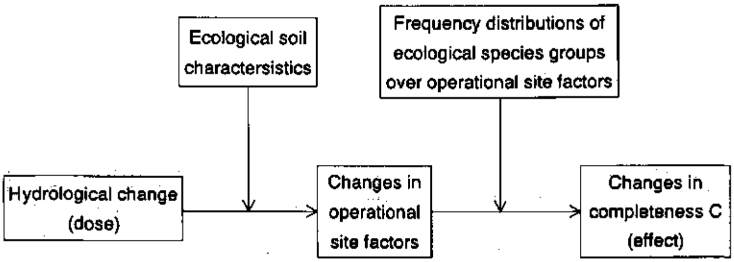

Figure 2.7: Calculation of the effect of a hydrological change (dose) on the completeness of the vegetation. Freely adapted after Runhaar *et al.* (1996b) and Runhaar *et al.* (1997c).

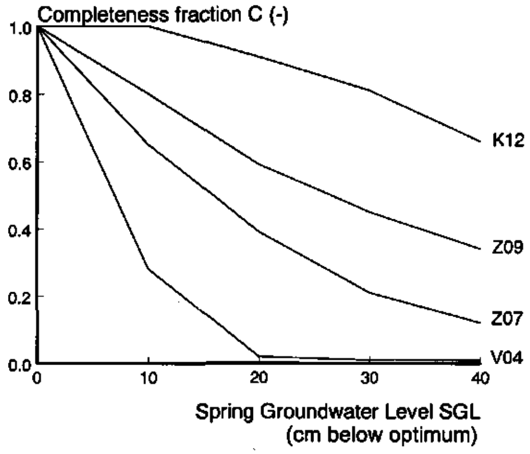

Figure 2.8: Predicted changes in C of ecotope group K21 on a lowering of the SGL for various ecological soil types: K12 - old strongly weathered clays; Z09 - loamy non-calcareous sands with thin topsoil; Z07 - non-calcareous sands with thin topsoil (not loamy); V04 - primary meso-eutrophic peat.

shows some dose-effect functions of ecotope group K21 (herbaceous vegetation on wet, nutrient-poor, acid soil) for a fall in the groundwater table ($\Delta SGL > 0$) on four ecological soil units. Similar functions were established for the other four doses (Δs, ΔSSL, ΔCl, ΔP).

From Fig. 2.8 it appears that ecotope group K21 is the most susceptible to a groundwater lowering on meso-eutrophic peats (V04). This is due to the fact that the effects of physiological desiccation, mineralization as well as of acidification are strong on this soil type. On old strongly weathered clays (K12), the susceptibility is limited because no mineralization effect is expected.

Construction of dose-effect functions

Starting point in the construction of the dose-effect functions has been a very well developed ecotope group ($C = 1$) under optimal hydrological conditions. For e.g. a wet ecotope group, the optimal SGL is near soil surface.

In the first step, data from field measurements and literature were used to estimate for each ecological soil unit the effect of a dose on the operational site factors of the ecotope system, characterized by:

Moisture regime:
- SGL (applied to terrestrial ecotope groups). In combination with the texture of the ecological soil unit, the SGL is indicative for the availability of soil moisture and oxygen;
- SSL in combination with the estimated depth of the surface water level (semi-terrestrial: A12, A17, A18, bA10);

Nutrient availability:

- N-mineralization (terrestrial). The mineralization rate depends on the amount and type of organic matter, which can be deduced from the ecological soil unit;
- P-content (semi-terrestrial);

Acidity:

- *pH* of the soil, which is dependent on the $CaCO_3$ content and the base saturation of the soil (terrestrial);

Salinity:

- Cl-content (semi-terrestrial).

On a particular ecological soil unit the dose ΔSGL, for instance, will result in a new SGL (moisture regime), a new N-mineralization rate (nutrient availability) and a new soil pH (acidity). The effect on the vegetation (step 2) was calculated with the help of frequency distributions of ecological plant species groups over operational site factors. Fig. 1.7 (Section 1.5.1), for example, shows the frequency distribution of 'wet' species over the SGL as a measure for moisture regime. Similar functions were also constructed for 'moist' species, but this time in relation to the soil texture. Also frequency distributions of 'nutrient-poor', 'moderately nutrient-rich' and 'very nutrient rich' species over N-mineralization (nutrient-availability) and of 'acid', 'neutral' and 'alkaline' species over *pH* (acidity) were made. Using these frequency distributions as response curves, the effect of changes in operational site factors (output step 1) on the ecological species groups could be calculated. Incorporating information about the species composition of the ecotope group, it was possible to combine these changes of ecological groups into a change in the completeness fraction C (Runhaar *et al.*, 1996b; Runhaar *et al.*, 1997c).

Van der Linden *et al.* (1992) were able to validate some of the dose-effect functions for fall in the groundwater table on observed effects, described in the literature. The predicted effects were largely consistent with the observations.

Application of the dose-effect functions in DEMNAT

Fig. 2.8 demonstrates that small changes in hydrology may have considerable effects on the vegetation. It is entirely clear that hydrological models - certainly on a national scale - are not able to produce results with an accuracy that matches the susceptibility of the vegetation, as expressed in the dose-effect functions. In flat areas for instance, the estimated accuracy of the computed SGL is a couple of dm's (Section 2.3.3), which is far beyond the range where major vegetational changes occur: there appears to be *a gap between the accuracy of hydrological models and the susceptibility of the vegetation.*

In DEMNAT this gap is provisionally closed by using *hydrological differences* as input variables, instead of absolute values. Hydrological models are more reliable in predicting differences than in predicting absolute values, since systematic errors are minimized when calculating a difference. DEMNAT's approach is demonstrated in Fig. 2.9A for a lowering of the SGL ($\Delta SGL > 0$). This figure shows

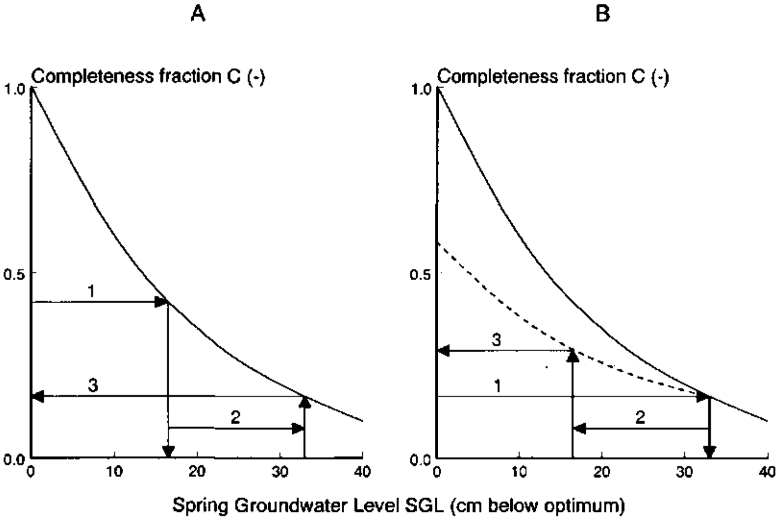

Figure 2.9: Use of the dose-effect functions: A - with a fall in the *SGL*; B - with a rise in the *SGL* (Witte, 1990). To begin with, the *SGL* before water management measures is estimated from the original completeness fraction (1). Then the predicted change in groundwater level, ΔSGL, is added to this initial value, resulting in a new *SGL* (2). And finally, the completeness fraction after water management measures is obtained (3). The dotted line in Fig. B accounts for an assumed hysteresis effect, which occurs when recovery takes place.

that the original completeness fraction is used to estimate the initial *SGL* of the vegetation, that is the *SGL* before any intervention in water management has taken place. Subsequently, the computed change in groundwater level, ΔSGL, is added to this initial value, resulting in a new *SGL* with a corresponding new completeness fraction C.

However, policy makers also want to be able to predict the recovery of nature as a result of favorable water management measures. As it takes time for an ecosystem to recover, *hysteresis factors* - which reduce the slope of the original dose-effect functions - have been temporarily introduced (Fig. 2.9B). These factors, which are based on professional judgement, depend both on the type of ecotope group and on the ecoseries type. They relate to the prediction term of 20 years.

For recovery, diaspores must be available, either by active dispersal from neighboring populations, or from the seedbank. As no national data are available about seed banks, DEMNAT demands a certain amount of characteristic plant species to be present in a km-square, before it computes any recovery.

2.3.6 Valuation of effects

If desired, the predicted changes in the completeness fractions of the ecotope groups can be expressed as changes of conservation values. Conservation valuation allows the outcomes for different ecotope groups to be combined, enabling results that are easy to interpret. Because the valuation of nature is a subjective affair, the prediction of effects on C is strictly separated from the assessment of these effects in terms of nature conservation. Moreover, the user of DEMNAT is able to change the valuation procedure to some extent. An extensive description of DEMNAT's conservation valuation procedure will be given in Chapter 5.

A *potential conservation value* V for each ecotope group was determined from the criteria 'national rarity' and 'international rarity' just once (Table 5.3, Section 5.2.4). This value refers to botanically very well developed km-squares of the ecotope group in question. The *actual value* of an ecotope group in a certain km-square is calculated by multiplying C with V. Adding all these values of ecotope groups within one km-square yields the conservation value of this km-square:

$$V_{\mathrm{km}} = \sum_{e=1}^{n_e} C_e V_e \tag{2.5}$$

where:

V_{km} = conservation value of the km-square [-]
e = index ecotope group [-]
n_e = total number of ecotope groups [-]
C = completeness fraction [-]
V = potential conservation value ecotope group [-]

The gain or loss of conservation values per km-square due to predicted changes in the completeness fraction is simply calculated as: $\sum_{e=1}^{n_e} \Delta C_e V_e$.

2.3.7 Examples of simulations with DEMNAT

DEMNAT-2 has been applied in a national policy analysis to prepare a document on drinking water and industrial water supply (Beugelink *et al.*, 1992). One of the scenarios analyzed concerned a 50% decrease of the drinking water extraction. In this section some results are given for a floristically well-investigated region in the Netherlands, the province of Utrecht (color Fig. 2.15). The color maps of Figs. 2.16 and 2.17 show the hydrological input of this scenario for DEMNAT. Fig. 2.16 gives the effect on the SGL per 500 m square. A substantial rise of the SGL can be expected in the ice-pushed ridge 'Utrechtse Heuvelrug', stretching roughly from SE to NW. Especially at the edge of this ridge intensities of upward seepage will increase (Fig. 2.17). The effect of this scenario on the influx of Rhine water is negligible (Pakes *et al.*, 1992). The ecological impacts of these hydrological changes were predicted with DEMNAT-2.1. Color Fig. 2.18 gives one of the

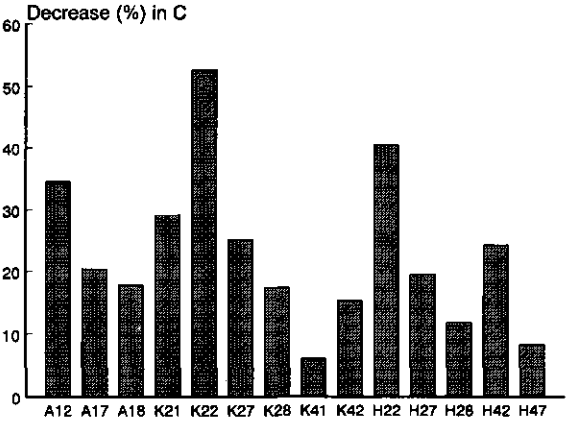

Figure 2.10: Predicted decrease in the average completeness fraction C in the Utrecht province as a result of a 10 cm fall in the groundwater level and the surface water level (ecotope groups that are extremely rare in this province are omitted).

results, the effect on the conservation value per km-square ΔV_{km}. A substantial increase in conservation value may be expected at the edge of the ice-pushed ridge. No effect will occur on the ridge itself. Some groundwater-dependent ecotope groups are present on the ridge (especially K21 and K41), but they only occur on soils that have a perched water table. As these soils cannot be influenced by water management, DEMNAT ignores these occurrences.

Color Fig. 2.19 shows another example of DEMNAT's output, which is the result of a completely fictitious scenario: an overall lowering of both SGL and SSL with 5, 10 and 20 cm, without any change in upward seepage or surface water quality. It needs no comment that nature suffers more as the water level decreases. Dramatic effects are predicted especially for the western part of Utrecht, where peaty soils are dominant. A highly affected region can be seen at the west side of the ice-pushed ridge. This region (Noorderpark) is well known for its mesotrophic fens, with a vegetation that is partly dependent on upward seepage (especially K22). Fig. 2.10 shows the effect of a 10 cm groundwater fall in spring on the average completeness fraction in Utrecht for all the ecotope groups. On average, the ecotope groups will decline by 23%, but for ecotope groups of wet, nutrient-poor and neutral sites (A12, K22, H22) the predicted decline is much more.

2.4 Discussion

2.4.1 Introduction

DEMNAT's power is determined by the fact that the model comprises all the elements that are needed for a prediction: geographical information, dose-effect functions and a valuating system. Another advantage is the relatively smooth connection with available hydrological models. In policy analyses DEMNAT has proved to be a helpful tool. As with any model, it can be questioned whether an expert on the subject would make better predictions. Though answers to this question are hard to validate, I think that DEMNAT has reached a state which makes the use of experts unnecessary.

In the next few chapters of this publication the geographical schematization and the valuation procedure of DEMNAT will be discussed in detail. In Section 2.4.2 I will discuss the dose-effect functions, especially the way they are dealt with within DEMNAT. In Section 2.4.3 I will account for the fact that DEMNAT uses plant species groups (viz. ecotope groups) as biotic variables instead of - as is the case with the majority of ecohydrological models - individual plant species. I will discuss some possibilities for the improvement of DEMNAT in Section 2.4.4.

2.4.2 Dose-effect functions

The dose-effect functions are used by DEMNAT in such a way that the initial groundwater level is deduced from the actual completeness fraction (Fig. 2.9). Of course this way of application is somewhat dubious, for the fact that an ecotope group is poorly developed might have more causes than just a sub-optimal groundwater level. Think for instance of poor vegetation management, or the atmospheric deposition of N.

But maybe the current approach is not so very bad. The dose-effect functions and the way they are currently interpreted show that differences between ecosystems in their susceptibility to environmental changes are dependent on:

1. the *type* of ecosystem - which is reflected by the general steepness of the dose-effect function;

and (most dose-effect functions being concave):

2. the *species richness* of the ecosystem (the higher C, the more susceptible), which is reflected by the slope of the dose-effect function at the initial SGL.

Both consequences seem justifiable and desirable. A sensitivity analysis showed that DEMNAT is rather insusceptible to the way the dose-effect functions are applied (Arts *et al.*, 1997), *as long as the results are used in a relative sense*, i.e. to compare various water management scenario's.

The way the dose-effect functions are applied has much to do with an important difference between DEMNAT and other ecohydrological models (WAFLO, ICHORS, MOVE), namely that *the initial site conditions* (such as moisture regime) *are derived from the vegetation itself.* In my opinion, far better information about actual site conditions may be obtained from the vegetation than from any other available database or model. For instance, ecotope group K21 indicates a wet, nutrient-poor and acid site, and DEMNAT assumes this site is present in a km-square whenever this is indicated by the flora-records in FLORBASE, *even if hydrological computations point at completely different circumstances.*

Furthermore, there is a difference between the functions statistical models (ICHORS, MOVE) use and those that DEMNAT uses for a recovery. Although the introduction of hysteresis factors is not a final solution, I do think the fact that it takes longer for a vegetation to recover than to deteriorate, should be taken into account.

2.4.3 Species or species groups?

In contrast to DEMNAT, most ecohydrological models make predictions for individual plant species. Plant species have the advantage of being clearly distinguishable units that are - unlike groups of plant species - nearly independent of a chosen classification. However, I hold the view that the use of plant species in a national model has more drawbacks.

Firstly, errors in a floristic databank directly affect the results when individual plant species are used. Witte & Van der Meijden (1993, 1995) - see Chapter 4 - were able to minimize these errors by taking groups of species, introducing a completeness class 'noise', and by 'gap-filling' per ecotope group.

Secondly, the dynamics in the appearance and disappearance of plant species exceeds the dynamics of ecological groups as a whole. This probably explains the moderate results of predictions for individual plant species (cf. Gremmen *et al.*, 1990, p. 152).

Thirdly, for easily manageable results, conservation valuation is a necessity. Conservation valuation on the basis of plant species gives an underestimation of species-poor ecosystems, such as bogs. Moreover, such a valuation results in giving a higher value to e.g. a heather with Nettles (*Urtica*; NL: Brandnetel) than a heather without. Therefore, it is better to valuate on the basis of the relative richness of species groups (as expressed in C). See Chapter 5.

In the fourth place, for a reliable prediction, the involvement of spatial variation in site conditions is a prerequisite. This variety is included in DEMNAT by means of site-diagrams (Section 2.3.4). These diagrams link the 18 ecotope groups of DEMNAT to the 435 ecoseries that appear to exist in reality. Hence, for the theoretical number of $18 \times 435 = 7830$ combinations of ecotope group and ecoseries, a 'frequency class' had to be estimated (e.g. Table 2.3). If we would use 700 plant species, like MOVE does, the number of possible combinations to be made would be $304,500$! Whether sufficient knowledge is available to implement an operation like this, may seriously be doubted.

Finally, national applicable response functions for individual plant species can only be derived from an extensive databank with a representative set of records of site characteristics and plant species. At present, such a databank is not available.

2.4.4 Possible improvements

DEMNAT may be compared to a large cathedral with some pillars out of granite and others out of clay. Clearly, improvements are possible and even required in order to obtain a model that is solid in all its components.

Presently, DEMNAT is not able to make predictions for certain regions because of a shortage of flora records in FLORBASE (Fig. 4.14). This problem was provisionally solved by Bleij & Witte (1996), who developed a procedure to merge the output of DEMNAT per km-square into large geographical units, such as provinces, meanwhile making up for shortcomings of FLORBASE. But the problem of inadequate flora data can only be solved substantially, if substantial amounts of plant records are added to FLORBASE. This is especially the case in areas where the present botanical survey falls short.

Substantial improvements of the dose-effect functions can only be established if more systematic data of vegetation together with site characteristics become available. At present, there are not enough reliable data available to validate the dose-effect functions (Gremmen *et al.*, 1990). Therefore, systematic monitoring programs running for a period of many years in carefully selected sites, are urgently needed.

In my opinion *DEMNAT currently is capable of giving a reasonable prediction of negative effects of national water management.* However, as for analyzing water management measures favorable to nature, I doubt if DEMNAT performs well enough. When predicting negative effects, the site conditions of ecosystems can be derived from floristic information: if there are enough indicator species of e.g. ecotope group K21 (herbaceous vegetation on wet, nutrient-poor, acid soil) in a km-square, it is reasonable to assume that a wet, nutrient-poor, and acid site is present in that km-square. But how do we acquire site information if substantial indicator species are lacking? The ecoseries do not provide enough information because important factors, such as vegetation management and nutrient-supply, have not been considered when mapping the ecoseries. An improvement might be the use of other geographical databases, for instance on nutrient supply and on land use (Thunnissen *et al.*, 1996). At present, a regional model is developed which strongly resembles DEMNAT, but which uses these geographical databases in order to get a clear spatial picture of site conditions (Meuleman *et al.*, 1996). A fair estimation of the actual depth of the groundwater table and the spatial variation in this depth, might be obtained by using - per ecoplot - data about the frequency distribution of the height of the soil surface (Spieksma & Schouwenaars, 1997).

As recovery and development of ecosystems is dependent on the availability of diaspores, also the sources of diaspores should be taken into account. Perhaps an indication of these sources may be obtained from recent and historical flora records (Runhaar & Witte, 1997).

It might be that, in the near future, DEMNAT will be submitted to a totally different kind of 'improvement'. At present, DEMNAT is only suitable for analyzing scenarios concerning water management, especially those scenarios that relate to desiccation. However, it may also become suitable for analyzing two other environmental stressors: acidification and eutrophication. As the current DEMNAT-version makes predictions via the operational factors 'soil moisture regime', 'nutrient availability' and 'acidity', this adjustment should be relatively simple.

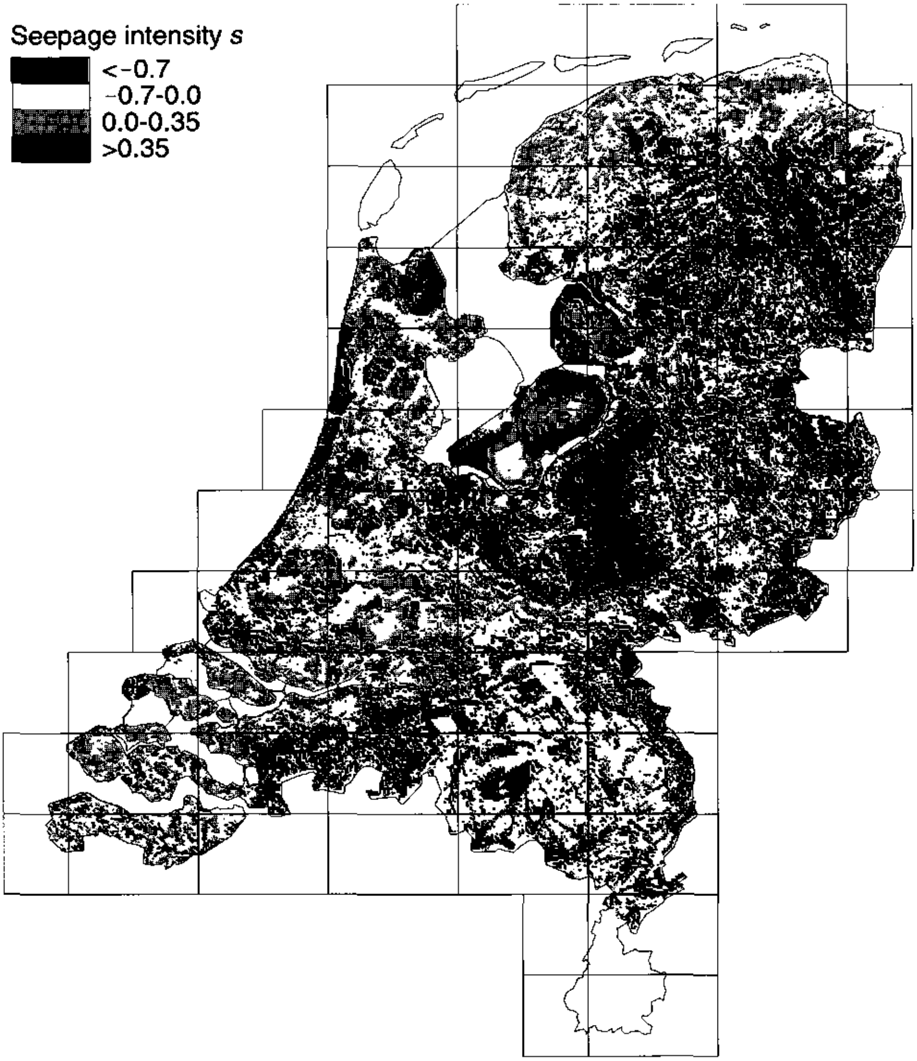

Figure 2.11: Seepage intensity (in mm d^{-1}) per 500 m square, computed with NAGROM/MONA (present situation). Positive values indicate upward seepage, negative values downward seepage. Courtesy of Vermulst *et al.* (1996).

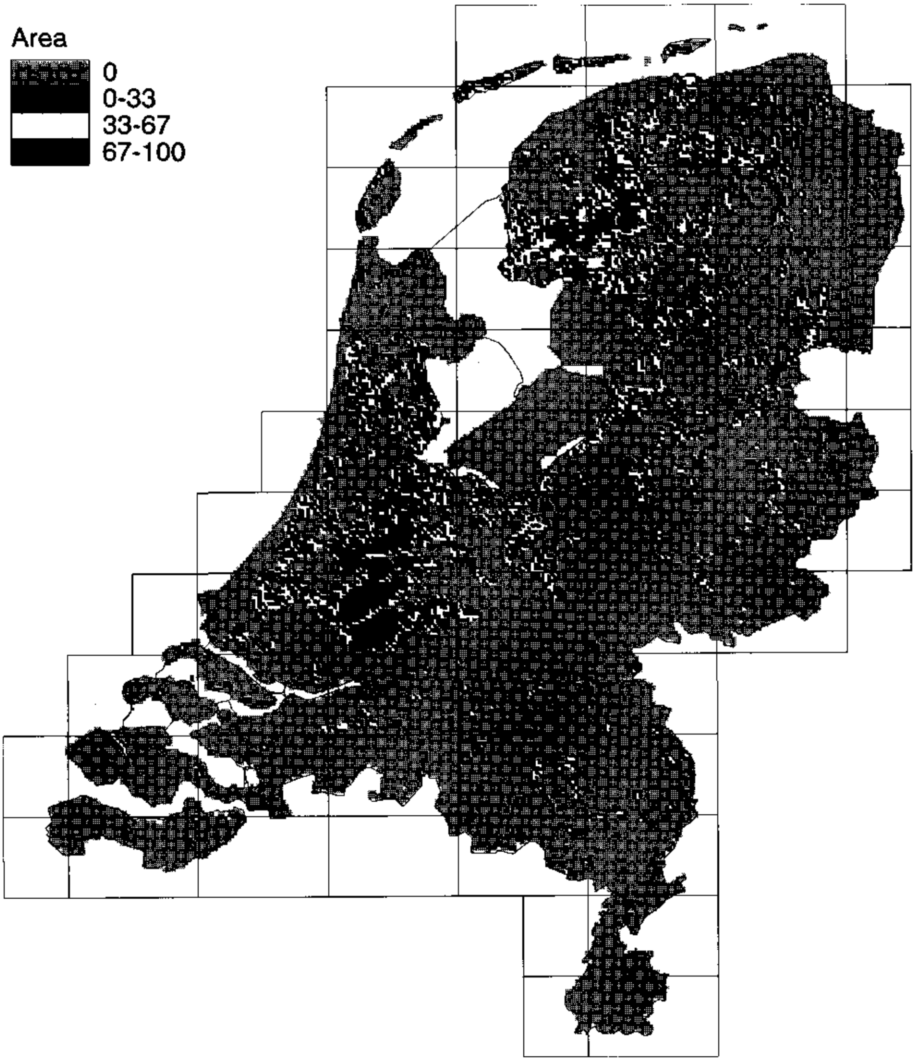

Figure 2.12: Distribution of groundwater table class GWT2 (shallow groundwater table), in ha per km-square. Courtesy of Klijn *et al.* (1996).

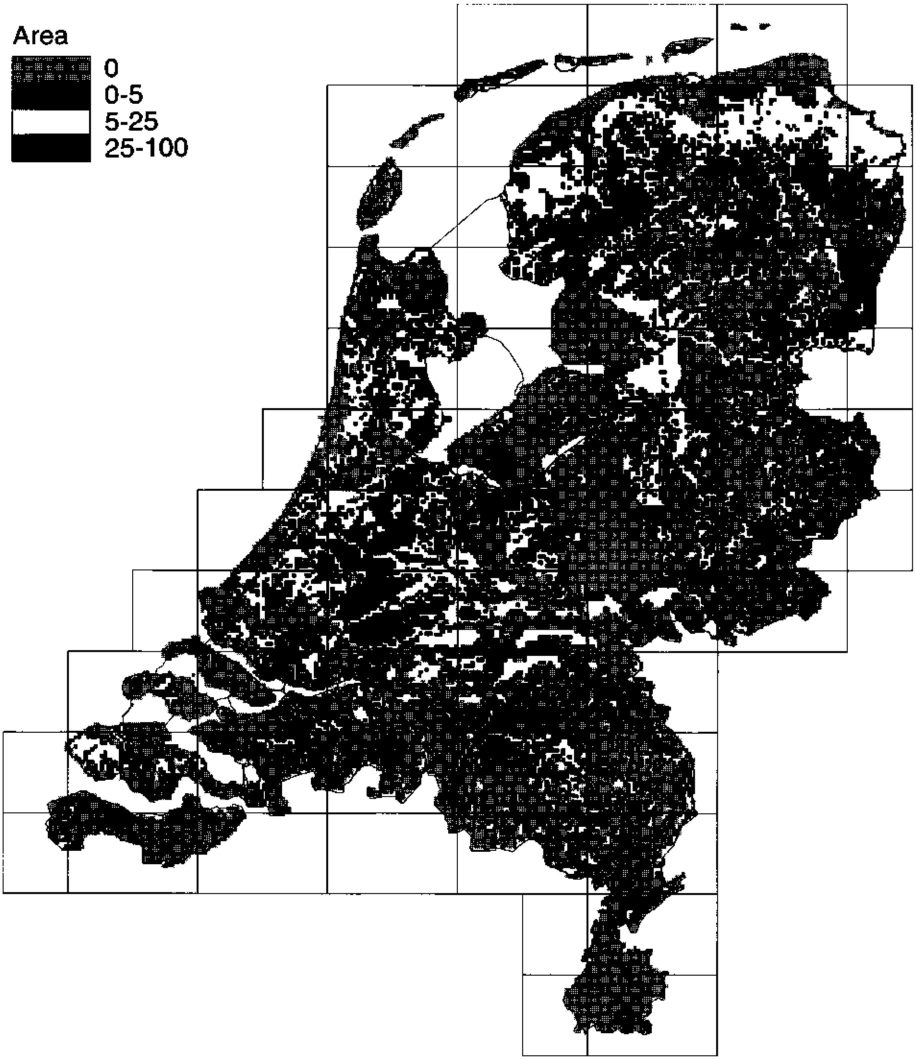

Figure 2.13: Distribution map of site type X27, in ha per km-square. Courtesy of Klijn *et al.* (1996).

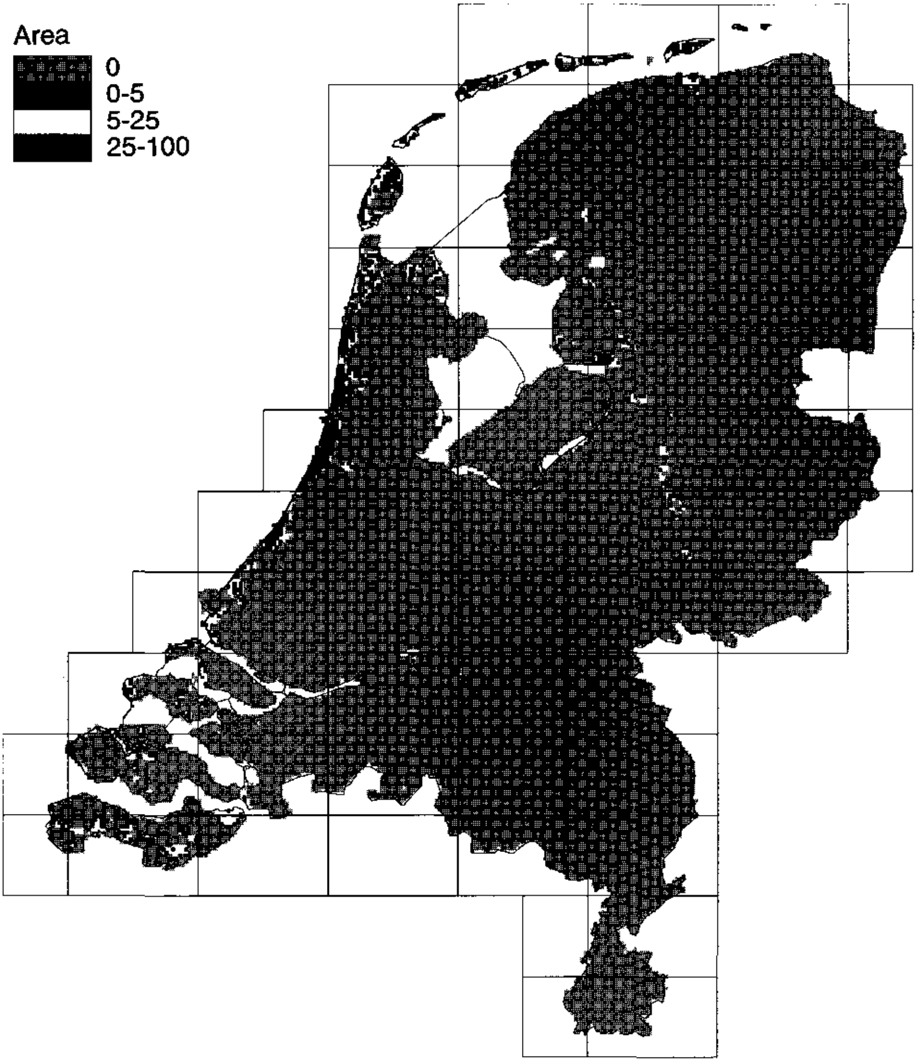

Figure 2.14: Distribution map of site type X63, in ha per km-square. Courtesy of Klijn *et al.* (1996).

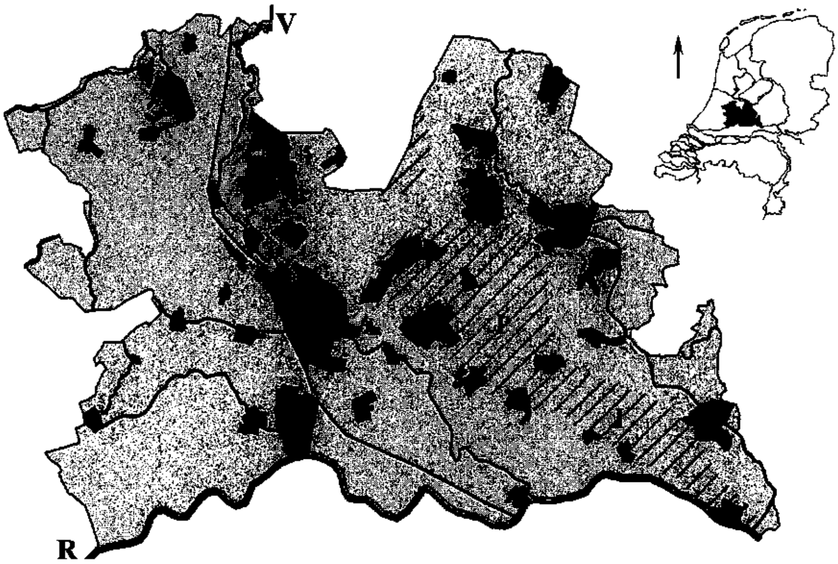

Figure 2.15: Province of Utrecht with: A - city of Amersfoort; I - ice-pushed ridge the 'Utrechtse Heuvelrug' and 'Het Gooi'; L - lake Loosdrecht; R - river Rhine; U - Utrecht city; V - river Vecht. Cities and villages are indicated by the color red and surface waters by the color blue.

Figure 2.16: Effect (in cm) on the average spring groundwater level *SGL* from a 50% decrease of the groundwater extraction for drinking water, computed with MOZART.

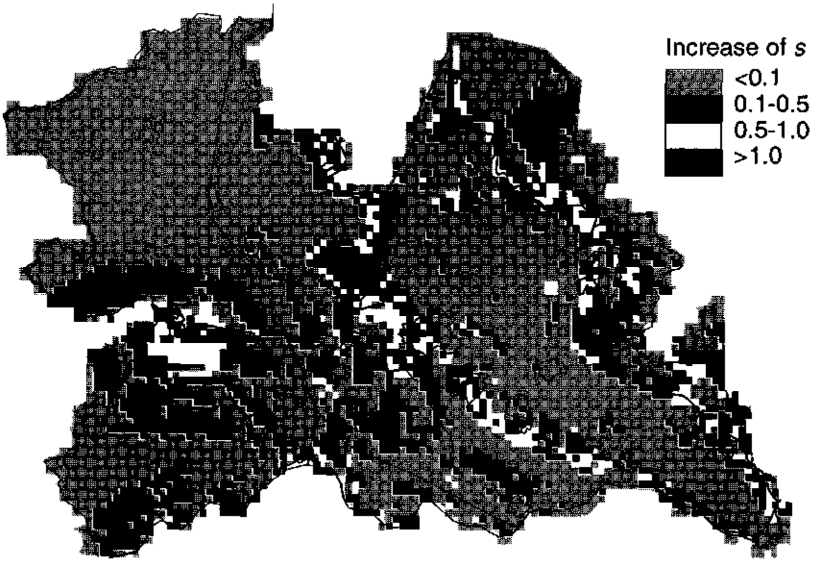

Figure 2.17: Rise of the intensity of upward seepage (in mm d^{-1}) due to a 50% decrease of the groundwater extraction for drinking water purposes, computed with NAGROM.

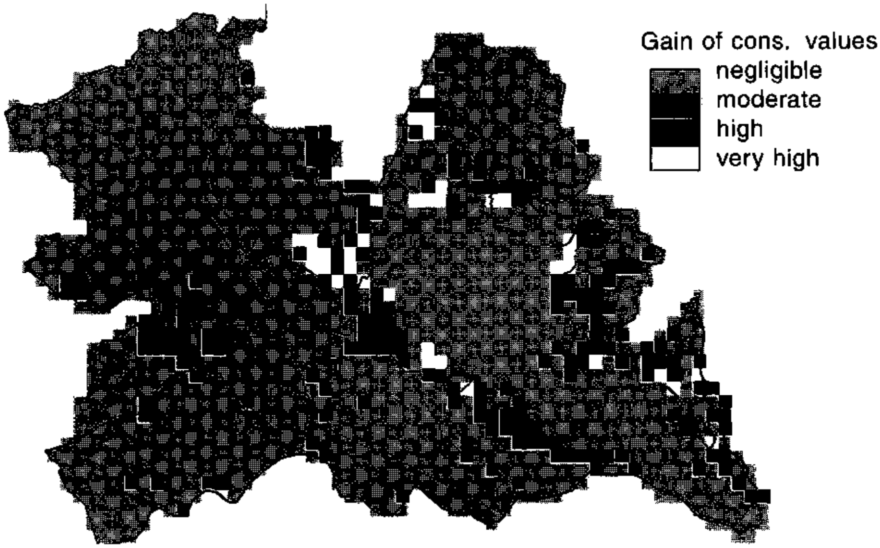

Figure 2.18: Predicted gain of conservation values as a result of a 50% decrease of the groundwater extraction for drinking water purposes.

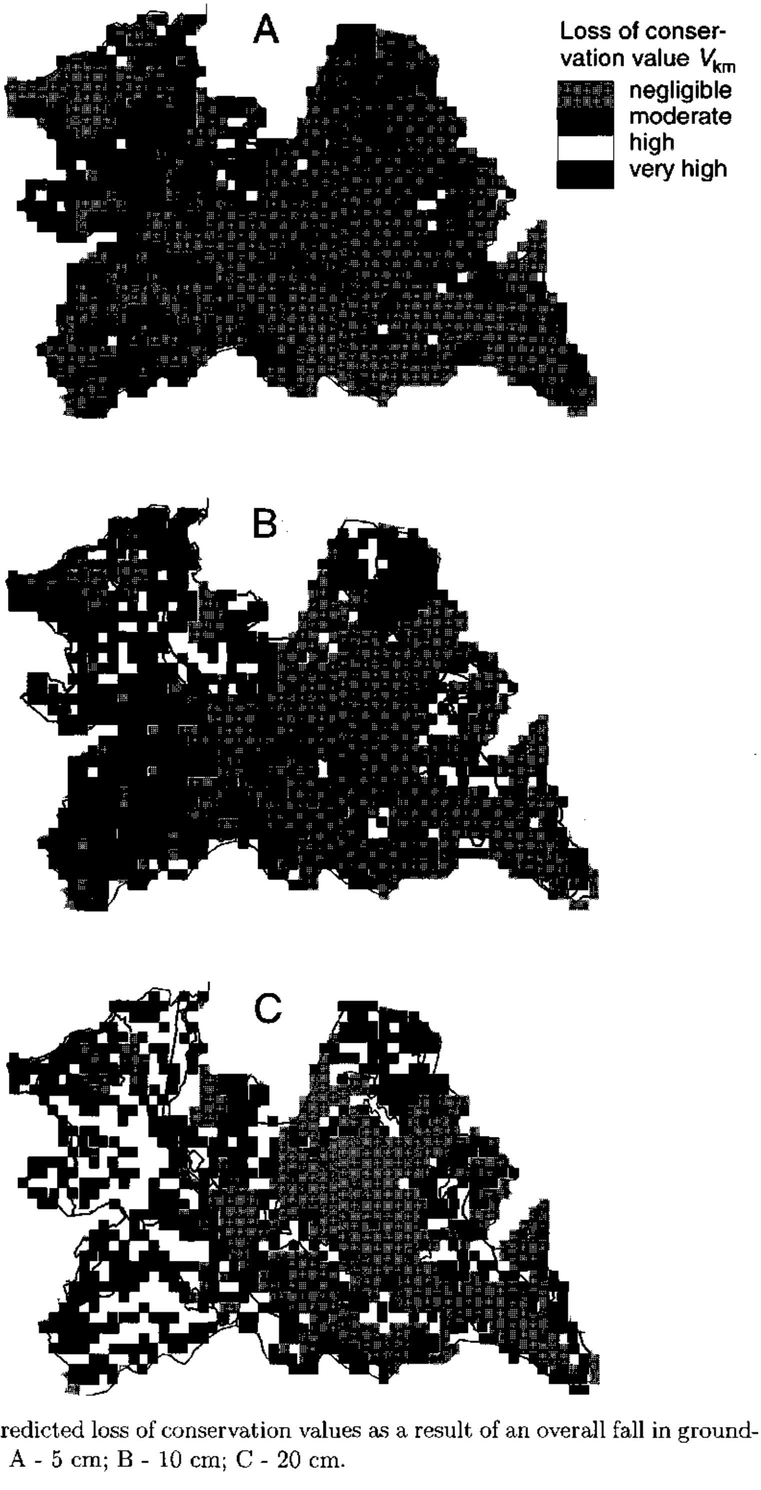

Figure 2.19: Predicted loss of conservation values as a result of an overall fall in groundwater level of: A - 5 cm; B - 10 cm; C - 20 cm.

Chapter 3

The descriptive capacity of the ecotope system

3.1 Problem definition

In order to carry out its nature conservation policy in the best possible way, the Dutch government wishes to be provided with data about the way nature is represented in its territory. FLORBASE (Section 1.4) nowadays provides these data in the most adequate way for it offers a full-scope and up-to-date picture in the field of botany. In Chapter 4, FLORBASE-2c is used for the making of some distribution maps of ecosystem types. These maps serve the ecohydrological model DEMNAT-2 (Chapter 2). The ecosystem types are based on the ecotope system that has been described in Section 1.6. It is self-evident that the ecotope system should be chosen since its classification units - the ecotope types - are defined according to the four site factors: salinity, moisture regime, acidity and nutrient availability. Precisely these factors are influenced by water management and therefore the system fits methods of ecohydrological impact assessment. Together with flora observations per kilometer square, the ecotope system has already proved to be very useful for water management analyses on a regional scale (De Heer *et al.*, 1996; Foppen *et al.*, 1996; Gieske *et al.*, 1996; Hoogeveen & Vermulst, 1997; Kloosterman *et al.*, 1996; Negenman *et al.*, 1996; Runhaar & Groen, 1993; Stuurman *et al.*, 1996; Vermeulen *et al.*, 1996).

An objection that is sometimes raised against the ecotope system (Everts *et al.*, 1986; Grootjans *et al.*, 1987) is that it is contrived, for reasons such as that it uses fixed boundary classes of site factors to define its types (see also Section 1.6.4). Species from an ecological group are said to occur less in combination with each other than species from a phytosociological group, which would make the system less suitable for describing the vegetation.

For policy analyses it is not only necessary that species are correctly ascribed to ecological groups, but also that proper use can be made of these ecological groups for describing a vegetation. Only if this is the case, after all, may botanically valuable vegetations be localized and may the effects of measures be judged upon. Therefore, in this chapter I will examine whether or not the ecotope system is suitable for describing the vegetation of the Netherlands with FLORBASE. To this end, the *descriptive capacity* of the ecotope system on a national level will be determined. In order to be able to give a sound judgement, the descriptive capacity of the phytosociological system according to Westhoff & Den Held (1969) will also be determined. The system of Westhoff & Den Held consists of a classification and description of all the phytosociological vegetation types of the Netherlands (Appendix C). Unlike the ecotope system - which was primarily designed for environmental impact assessment - the phytosociological system was compiled to describe the vegetation as adequately as possible. So there is no doubt that the latter should be taken as a standard of comparison.

I have not used the latest phytosociological descriptions of Dutch vegetation types by Schaminée *et al.* (1995) because the last volume of these descriptions was issued too late for this purpose, namely in 1998.

Most of this chapter was been published before by Witte *et al.* (1996).

3.2 Method

3.2.1 Principle

Species are ascribed to classification units in both the phytosociological system and the ecotope system. When species are correctly assigned to a certain unit, they may be expected to occur frequently together in km-squares. Hence, species from, for example, association 25Aa1 (*Crepido-Juncetum acutiflori*) should occur relatively frequently in the same kilometer squares; roughly speaking, their distribution patterns should have more common denominators than the distribution patterns of species from other associations. So species from one classification unit (type) should frequently occur together spatially; in other words - they should have a relatively high *internal correlation*. But it is also important that species in one unit are distinguishable from species in other units, i.e. that they have a low *external correlation*. A low external correlation, which comes down to a large *distinguishing capacity*, is needed to describe the diversity of the plant cover of the Netherlands. If a classification system would specialize in, for example, nutrient-poor habitats, then high internal correlations would meet the expectations. However, the external correlations would also be high and the system would only partially describe the vegetation of the Netherlands. *The descriptive capacity of both systems can therefore be measured by internal, as well as by the external correlations of the species.* In this chapter these correlations are derived from FLORBASE-2c's distribution data of species.

3.2.2 Selection of species and kilometer squares

Data stored in FLORBASE come from numerous sources, including observations from provinces, flora working groups, and individuals (Appendix A). This explains

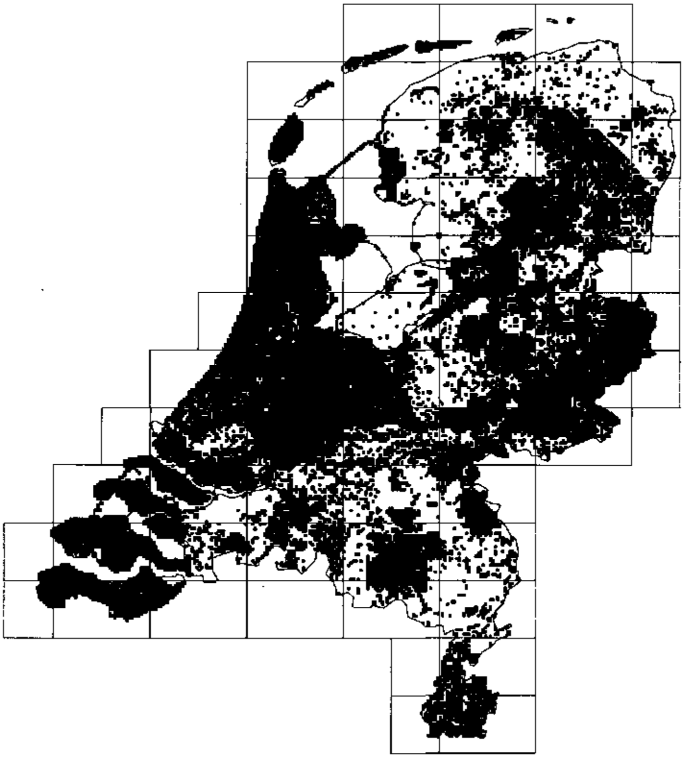

Figure 3.1: Grid-cells (1 × 1 km) that were used for the analyses. Each cell contains at least 50 selected species in FLORBASE-2c.

why the quality of the inventory of the km-squares in the database is variable. For the purpose of these calculations only fairly well investigated km-squares are selected, i.e. km-squares in which at least 50 plant species occur. Fig. 3.1 shows that mainly the provinces of Noord- and Zuid-Holland, Zeeland, Drenthe, Overijssel, Gelderland and Utrecht are selected for the comparison (Fig. 4.1 shows where these provinces are located).

Before any calculations can be made two selections are required. Firstly, as mentioned in Section 1.4.3, only wild vascular plants are used in the analysis. Besides, certain species such as trees and shrubs are excluded. Secondly, for a comparison, we must limit our choice to species with an unequivocal diagnostic value, that is, to species that exclusively belong to one classification unit. In the ecotope system a species like *Andromeda polifolia* (UK: Bog-rosemary; NL: Lavendelhei) is usable because it belongs exclusively to ecotope type P21. I will call such an unambiguously assigned species, an *ecotope character species*. On the other hand, *Trichoporum cespitosum* (UK: Deergrass; NL: Veenbies), for example, is excluded from the analysis of the ecotope system because it belongs to more than one ecotope type (i.e. G21 and G41). In the analysis of the phytosociological system, character species are used because each character species belongs to one specific phytosociological vegetation unit. Note: the species that are selected in this way need not necessarily be exclusively diagnostic. They simply have such a strong preference for one particular classification unit, that they have not been ascribed to any other units.

3.2.3 Correlation calculation

For each pair of diagnostic species i, j the correlation coefficient can be calculated from FLORBASE as (Johnson & Wichern, 1988, p. 552):

$$r = \frac{ad - bc}{\sqrt{(a+b)(c+d)(a+c)(b+d)}} \tag{3.1}$$

where:

r = correlation coefficient of species i and j
a = number of km-squares with i and j
b = number of km-squares with i only
c = number of km-squares with j only
d = number of km-squares without i and without j

An example is shown in Table 3.1, where the correlation coefficients of ecotope type P21 are included and also those between the ecotope types P21 and R28. As was to be expected the correlations within P21 are relatively high, whereas those between P21 and R28 are low. From the results we may calculate the *internal*

Table 3.1: Results of correlation calculations. The mutual correlation coefficients r of species from ecotope type P21, and the coefficients between species from P21 and R28 are shown. Species names and numbers according to Van der Meijden (1996).

Ecotope type	Species name	Species number	Species number			
			417	777	1068	1069
P21	Drosera intermedia	417	1.00	0.40	0.67	0.64
	Lycopodiella inundata	777	0.40	1.00	0.33	0.34
	Rhynchospora alba	1068	0.67	0.33	1.00	0.66
	Rhynchospora fusca	1069	0.64	0.34	0.66	1.00
R28	Epilobium parviflorum	457	-0.07	-0.02	-0.06	-0.05
	Euphorbia palustris	496	-0.01	0.00	-0.01	-0.01
	Inula britannica	662	-0.02	-0.01	-0.02	-0.01
	Senecio fluviatilis	1186	-0.02	-0.01	-0.01	-0.01

correlation of a unit as an arithmetic mean of the species belonging to this unit. Hence, from Table 3.1 it would follow that the internal correlation of P21 is 0.51. In the same way we may calculate the *external correlation* between two units, which, according to the table, is −0.02 for the combination P21-R28. A mean may be determined, once more, from the external correlations of a certain unit with all other units. This mean shows how one particular unit may be discerned from all other units together.

As FLORBASE data are stored in km-squares, the results of these calculations will be affected by 'noise': the spatial correlation between species in km-squares may be strong although they grow on different sites (for instance dry and wet heath terrains). In comparison with terrain data this noise might cause correlations that are too high.

3.3 Comparison between ecotope types and alliances

3.3.1 Introduction

Disregarding the additional characteristics, Runhaar *et al.* (1987) distinguished 70 ecotope types. Disregarding the epiphytic communities of mosses and lichens, Westhoff & Den Held distinguished 87 alliances and 193 associations. The fact that there are about as many ecotope types as there are alliances calls for a comparison between these two types of units.

As was mentioned before, only species with a clear diagnostic value are used in the comparison. Moreover, only units comprising at least three diagnostic species will be included in the calculation, because a low number of species will not provide a reliable statement about the unit. These conditions leave only 40 ecotope types and 42 alliances for the comparison, comprising 380 and 276 diagnostic species, respectively. Analysis based on the indicator values of Ellenberg (1991) shows a similar distribution of the selected ecotope types and alliances over the four site factors: salinity, moisture regime, nutrient availability and acidity (Fig. 3.2). A comparison is therefore justifiable. Fortunately, Fig. 3.2 also illustrates that the selected units give a fair representation of the whole of the Dutch flora.

3.3.2 Highest correlated units

In Table 3.2 the selected ecotope types are arranged in a descending order of internal correlation. This table also gives the ecotope type with which each ecotope type has the highest external correlation, i.e., the type with which it most frequently occurs in a km-square. A similar table for the alliances is given in Table 3.3.

The highest correlated pairs in Table 3.2 are often ecotope types from one and the same site, the only difference being their vegetation structure. For example,

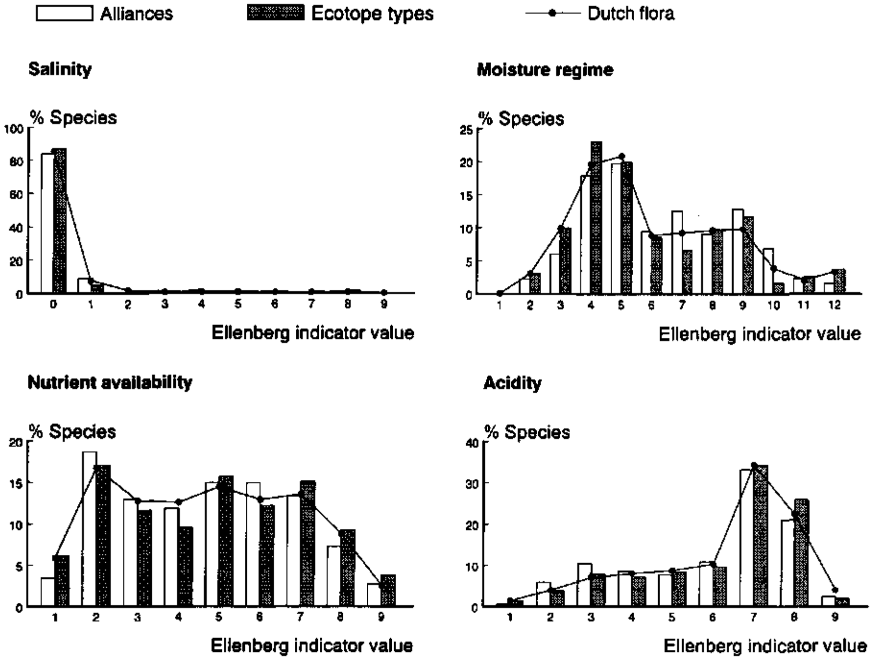

Figure 3.2: Frequency distributions of the selected character species of ecotope types and alliances over Ellenberg's indicator values of salinity, moisture regime, nutrient availability and acidity. As a reference, the frequency distributions of all the species of the Dutch flora are also shown.

pair 1 comprises types (P21-G21) characteristic of wet, nutrient-poor and acidic conditions (site type X21). Of the 34 possibilities that an ecotope type has of 'choosing a partner' from the same site, 20 have been used, see Table 3.2. In other cases often a very closely allied ecotope type is chosen as a partner; e.g. No. 9 (P22-P21) and No. 5 (W18-W17; V18 is not in the selection). Four wood types appear to have been linked to a wood type from a different site even though an ecotope type from the same site was available (Nos. 14, 20, 21, 38). In this way H27 is paired with H42 (No. 14) whereas G27 or R27 could also have been chosen. It seems that the factor 'wood environment' (shade, leaf fall) in the species composition of the vegetation dominates the site factors of moisture regime, nutrient availability and acidity. Another possible explanation is that because of the grid-cell size of 1 km^2 ecotope types are linked together that have a different site but on the other hand spatially occur close to each other in the landscape. Clear examples probably resulting from apparent correlations are the combinations bP40-zG20 (No. 11), G22-V17 (No. 22), G42-G23 (No. 30) and bG40-zG20 (No. 39). In each of these four cases, there is a partner from the same site type available, but instead a partner from a site type has been chosen that deviates

Table 3.2: Pairs of highest correlated ecotope types, placed in a descending order of internal correlation. No. = sequence number, $e_{\max}$ = highest correlated type. See Table 1.3 for an explanation of the ecotope type codes. Example: ecotope type zG20 has the second highest internal correlation and has the highest external correlation with zP20.

No.	Eco. type	$e_{\max}$	No.	Eco. type	$e_{\max}$	No.	Eco. type	$e_{\max}$
1	P21	G21	15	H63	G63	29	R48	P48
2	zG20	zP20	16	P63	G63	30	G42	G23
3	bP60	bP40	17	W17	V17	31	R47	G67
4	G21	P21	18	P48	G48	32	R28	P28
5	W18	W17	19	P28	R28	33	G48	P48
6	zP20	zG20	20	H47	H27	34	W12	P21
7	G23	bP60	21	H42	H27	35	G47	G48
8	G43	H43	22	G22	V17	36	P47	G43
9	P22	P21	23	G27	V17	37	G67	P67
10	G63	H63	24	P62	G62	38	H62	H27
11	bP40	zG20	25	H41	G21	39	bG40	zG20
12	V17	W17	26	bG20	zG20	40	R27	V17
13	H43	G43	27	P67	G67			
14	H27	H42	28	G62	P62			

at least two classification classes from the ecotope type in question (e.g.: bP40 deviates in salinity as well as in moisture regime from its partner zG20; bG40 would have given the perfect match with bP40).

Anyhow, as already mentioned, Table 3.2 mainly shows that ecotope types from the same or closely allied sites are well correlated with each other. Consequently, we may conclude that correlation calculations using FLORBASE-2c data produce meaningful results, in spite of the drawback of noise. Besides, it appears that *on a national scale the ecotope system has descriptive capacity.*

A closer look at the habitat descriptions in Westhoff & Den-Held (1969) discloses that also in Table 3.3 ecologically allied units are grouped.

The phytosociological system is hierarchical: associations are grouped into alliances; and these, in turn, into orders; and these again into classes. Some investigators ascribe a spatial significance to this hierarchy. Mueller-Dombois & Ellenberg (1974), for example, believe that the association level can be shown on a map on a scale of $1:5,000$, whereas the alliance level is chartable on a scale of $1:10^5$ to $1:10^6$. In Table 3.3 little of this possible spatial aspect of the hierarchy is detectable: of the 18 possibilities that two alliances have of being grouped under the same order, 4 are used (Nos. 3, 16, 29, 35), and of the 32 possibilities for two associations to be grouped in one class, 7 are successful (Nos. 3, 8, 13, 15, 16, 29, 35).

Naturally, also in Table 3.3 there are combinations probably resulting from aparant correlations. For instance: 20Bc is paired with 15Ab (No. 5), wereas two alliances

Table 3.3: Pairs of highest correlated alliances, placed in a descending order of internal correlation. *No.* = sequence number, A_{max} = highest correlated alliance. Explanation: Table 3.2.

No.	Alliance		A_{max}	
1	15Ab	Ammophilion borealis	20Bc	Galio-Koelerion
2	19Bb	Oenanthion aquaticae	5Ab	Nymphaeion
3	24Aa	Puccinellion maritimae	24Ab	Armerion maritimae
4	37Aa	Quercion robori-petreaea	18Aa	Epilobion angustifolii
5	20Bc	Galio-Koelerion	15Ab	Ammophilion borealis
6	5Ab	Nymphaeion	19Bb	Oenanthion aquaticae
7	12Bb	Polygono-Coronopion	13Ab	Aphanion
8	19Ac	Cicution virosae	19Ba	Phragmition (communis)
9	18Aa	Epilobion angustifolii	37Aa	Quercion robori-petreaea
10	25Ab	Filipendulion	19Bb	Oenanthion aquaticae
11	27Aa	Caricion curto-nigrae	25Ac	Junco (subuliflori)-Molinion
12	30Ba	Calluno-Genistion pilosae	20Ba	Thero-Airion
13	19Ca	Magnocariceon	19Ac	Cicution virosae
14	13Ab	Aphanion	12Bb	Polygono-Coronopion
15	19Ba	Phragmition (communis)	19Ac	Cicution virosae
16	25Aa	Calthion palustris	25Ab	Filipendulion
17	27Ba	Caricion davallianae	15Ab	Ammophilion borealis
18	20Ba	Thero-Airion	18Aa	Epilobion angustifolii
19	34Ab	Berberidion	20Bc	Galio-Koelerion
20	25Ac	Junco (subuliflori)-Molinion	37Aa	Quercion robori-petreaea
21	12Bc	Helminthion echiodis	24Aa	Puccinellion maritimae
22	6Aa	Littorellion uniflorae	5Ca	Potamion graminei
23	25Ba	Arrhenatherion elatoiris	12Bb	Polygono-Coronopion
24	11Ab	Chenopodium fluviatile	19Bb	Oenanthion aquaticae
25	38Aa	Alno-Padion	37Aa	Quercion robori-petreaea
26	23Aa	Saginion maritimae	24Aa	Puccinellion maritimae
27	21Aa	Mesobromion	34Ab	Berberidion
28	13Aa	Arnoseridion	37Aa	Quercion robori-petreaea
29	38Ab	Carpinion betuli	38Aa	Alno-Padion
30	12Ba	Sisymbrion	13Ab	Aphanion
31	12Aa	Polygono-Chenopodion	13Ab	Aphanion
32	10Aa	Nanocyperion flavescentis	18Aa	Epilobion angustifolii
33	16Ab	Agropyro-Rumicion crispi	12Bb	Polygono-Coronopion
34	12Bd	Onopordion acanthii	34Ab	Berberidion
35	24Ab	Armerion maritimae	24Aa	Puccinellion maritimae
36	17Ba	Senecionion fluviatilis	20Bb	Sedo-cerastion
37	20Bb	Sedo-cerastion	17Ba	Senecionion fluviatilis
38	30Aa	Violion caninae	37Aa	Quercion robori-petreaea
39	5Ca	Potamion graminei	6Aa	Littorellion uniflorae
40	17Aa	Arction	13Ab	Aphanion
41	13Ba	Caucalidion lappulae	21Aa	Mesobromion
42	5Bc	Callitricho-Batrachion	12Bc	Helminthion echiodis

from the same order were available (i.e. 20Ba and 20Bb). Here, alliances are concerned that occur spatially close to each other in the landscape: in lime-rich dune grasslands (20Bc *Galio-Koelerion*) and on the dunes along the coastal ridge (15Ab *Ammophilion borealis*). Another example of such a 'misfit' is the combination 12Bc-24Aa (No. 21; 12Ba, 12Bb and 12Bd were also available).

3.3.3 Correlation diagrams

In Fig. 3.3 the mean external correlations of the alliances and the ecotope types are plotted against the internal correlations. Remarkably, the internal correlations are largely low. Interestingly, it seems that plants take little notice of man-made classification units. Nevertheless, the figure shows that - to the exception of one ecotope type and one alliance (i.e. R27 and 05Bc) - the internal correlations are higher than the average external correlations; a reassuring thought for the classifiers.

The average values and standard errors within and between the units are included in Fig. 3.3. For comparison, these figures are also shown for two artificial divisions; a random division and a geographically optimum division. The optimum division is based on the highest possible internal correlations in FLORBASE and is not ecologically optimal (though it is doubtless that there are all kinds of underlying ecological factors). Appendix D shows how these divisions have been achieved.

On average, the internal correlation of the alliances is slightly higher than that of the ecotope types, but the difference is not of statistical significance ($P > 0.05$;

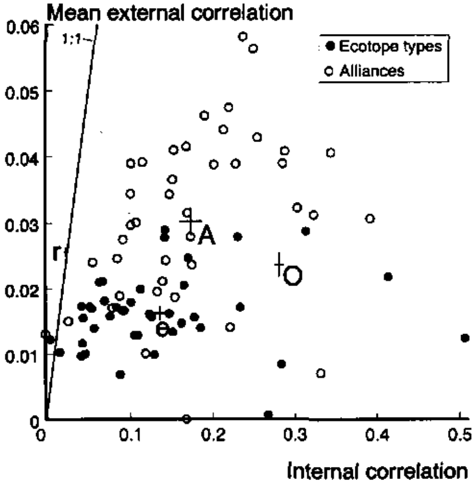

Figure 3.3: Mean external correlations versus internal correlations of ecotope types (closed circles) as well as alliances (open circles). The crosses show the means and standard errors: A = alliances, e = ecotope types, r = random types, O = optimal types.

t-test of difference between averages; Parker, 1979, p. 21-22). A higher internal correlation of the alliances could be expected because the selection of the alliance character species has been somewhat stricter (the average number of diagnostic species per alliance is 6.6 and per ecotope type 9.5).

On average, the external correlation in the ecotope system is significantly lower than that in the system of Westhoff & Den Held ($P < 0.001$). Therefore, I conclude that, at the level of ecotope types and alliances, *the ecotope system has a greater distinguishing capacity than the system of Westhoff & Den Held.*

The average external correlations in Fig. 3.3 illustrate how units are spatially distinguishable from all other units put together. The following comparison shows the distinction between units that are spatially closely correlated. The method of comparison will show much resemblance with the 'corellogram' ('variogram'), which is for example used in geostatistics.

For each classification unit, the unit with the highest external correlation is given in Tables 3.2 and 3.3. The mean external correlation of all these highest correlated pairs ('interval' 1) is 0.121 for the ecotope types and 0.171 for the alliances. If we calculate in the same way the mean external correlation for the units with which the second highest correlation is reached (interval 2), we arrive at 0.085 and 0.137 for the ecotope types and alliances, respectively. The values for several of such *intervals of similarity* can be found in Table 3.4, where interval 0 gives the mean correlation within the unit, hence the *internal correlation.* The speed by which the correlations decrease as the interval increases tells us much about the distinguishing capacity of the division, especially when compared with the internal correlation. Therefore, I use the decrease in the mean external correlation as a percentage of the correlation at interval 0 as a measure for *spatial distinction*:

$$d = 100\frac{r_0 - r_e}{r_0} \tag{3.2}$$

where:

d = spatial distinction
r_0 = internal correlation coefficient
r_e = mean external correlation coefficient

The last two columns of Table 3.4 illustrate the results of these calculations.

In Fig. 3.4 the obtained spatial distinction is plotted against the interval. In order to make a fair and square comparison - despite the differences in the number of units involved in the analysis (40 ecotope types, 42 alliances) - the intervals of the ecotope types and the alliances are mutually adjusted. This figure also includes the curve of the random division (negative values are omitted) and the curve of the geographical optimal division.

From the figure it appears that both systems perform better than the random division. The ecotope system approaches the optimal division and again performs

Table 3.4: Computation with Eq. (3.2) of the spatial distinction d of both ecotope types and alliances from the external correlation r_e at several intervals of similarity.

Interval	Mean external correlation r_e		Spatial distinction d (%)	
	Ecotope types	Alliances	Ecotope types	Alliances
0	0.136	0.174	0	0
1	0.121	0.171	11	2
2	0.085	0.137	38	21
3	0.072	0.122	47	30
4	0.061	0.110	55	37

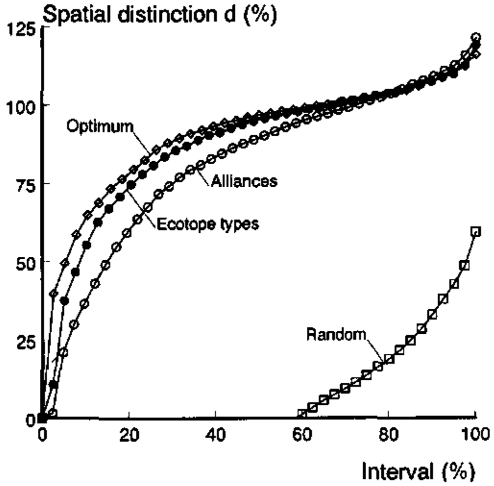

Figure 3.4: Spatial distinction d versus interval of similarity of both ecotope types and alliances. As a reference, also the curves of the random division and the optimal division are shown. The more the units differ in the distribution of their diagnostic species (increasing interval), the greater the spatial distinction between them. A steep curve over a short interval indicates that the division has a high distinguishing capacity.

better than the phytosociological system. The outcome of this comparison once more justifies the conclusion that - at the level of alliances - *the ecotope system has a greater distinguishing capacity than the phytosociological system of Westhoff & Den Held.*

3.4 Comparison between ecotope types and associations

Once again, a minimum of 3 diagnostic species per unit is taken, leaving us with 200 character species divided among 45 associations (the number of ecotope types and accompanying diagnostic species remain the same: 40 and 380). These associations - according to Ellenberg's indicator values - also cover similar habitats as the ecotope types investigated.

Table 3.5 gives the highest correlated pairs of associations in descending order of internal correlation. Six pairs consist of associations from the same alliance (Nos. 3, 17, 21, 37, 42, 45) whereas there are 24 possibilities for forming such pairs. With this selection, the distinguishing capacity of the ecotope system is again significantly higher than that of the system of Westhoff & Den Held (Figs. 3.5 and 3.6).

Conclusions from these results should be drawn with care, for the selected associations (193) amply outnumber the ecotope types (70). Besides, the selection

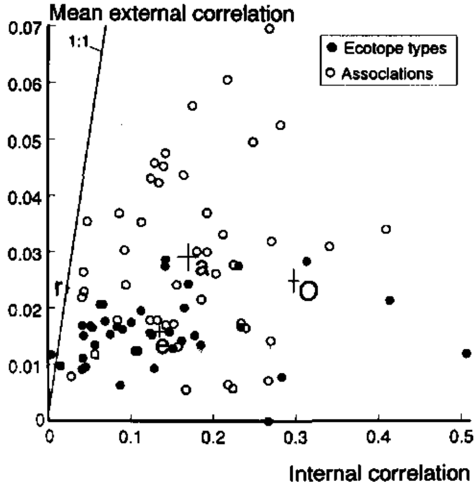

Figure 3.5: Mean external correlations versus internal correlations of ecotope types (closed circles) as well as associations (open circles). The crosses show the means and standard errors: a = associations, e = ecotope types, r = random types, O = optimal types.

Table 3.5: Pairs of highest correlated associations, placed in a descending order of internal correlation. No. = sequence number, $a_{\max}$ = highest correlated association. Explanation: Table 3.2.

No.	Association		$a_{\max}$	
1	20Bc2	Tortulo-Phleetum arenarii	12Bd2	Echio-Verbascetum
2	10Aa6	Eleocharitetum soloniensis	21Aa2	Medicagini-Avenetum pubesc.
3	12Aa5	Echinochloo-Setarietum	12Aa4	Oxalido-Chenopodietum polyspermi subatlanticum
4	11Aa1	Polygono-Bidentetum	12Aa4	Oxalido-Chenopodietum polyspermi subatlanticum
5	10Aa3	Centaurio-Sagin. moniliformis	20Bc2	Tortulo-Phleetum arenarii
6	17Ab1	Alliario-Chaerophylletum temuli	17Aa1	Tanaceto-Artemisietum
7	9Bb1	Atriplicetum sabulosae	23Aa1	Sagino maritimae-Cochlaerietum danicae
8	12Bd2	Echio-Verbascetum	20Bc2	Tortulo-Phleetum arenarii
9	38Aa1	Carici remotae-Fraxinetum	37Aa2	Fago-Quercetum
10	21Aa1	Koelerio-Gentianetum	38Ab1	Stellario-Carpinetum
11	11Aa2	Ranunculo-Rumicetum maritimi	16Ab2	Rumici-Alopecuretum geniculati
12	25Ac1	Cirsio-Molinietum	35Aa1	Carici elongatae-Alnetum
13	17Aa1	Tanaceto-Artemisietum	17Ab1	Alliario-Chaerophylletum temuli
14	10Aa1	Cicendietum filiformis	25Ac1	Cirsio-Molinietum
15	21Aa2	Medicagini-Avenetum pubesc.	10Aa6	Eleocharitetum soloniensis
16	12Aa3	Veronico-Lamietum hybridi	12Ba1	Chenopodio-Utricetum urentis
17	20Bc6	Anthyllido-Silenetum nutandis	20Bc2	Tortulo-Phleetum arenarii
18	38Aa5	Violo odoratae-Ulmetum	17Ab1	Alliario-Chaerophylletum temuli
19	35Aa1	Carici elongatae-Alnetum	19Ca5	Caricetum acuto-vesicariae
20	25Ab1	Valeriano-Filipenduletum	19Ca5	Caricetum acuto-vesicariae
21	12Aa4	Oxalido-Chenopodietum polyspermi subatlanticum	12Aa5	Echinochloo-Setarietum
22	12Bc1	Medicagini-Toriletum nodosae	12Aa3	Veronico-Lamietum hybridi
23	11Ab1	Malachio-Bidentetum fluviatile	11Aa1	Polygono-Bidentetum
24	7Ab3	Asplenietum trichomano-rutae-murariae	17Ab1	Alliario-Chaerophylletum temuli
25	13Ba1	Linarietum spuriae	12Aa2	Mercuriali-Fumarietum
26	19Ca5	Caricetum acuto-vesicariae	25Ab1	Valeriano-Filipenduletum
27	12Aa2	Mercuriali-Fumarietum	17Ab1	Alliario-Chaerophylletum temuli
28	23Aa1	Sagino maritimae-Cochlaer. d.	20Bc2	Tortulo-Phleetum arenarii
29	16Ab2	Rumici-Alopecuretum geniculati	11Aa2	Ranunculo-Rumicetum maritimi
30	25Ba1	Arrhenatheretum elatoiris	21Aa2	Medicagini-Avenetum pubesc.
31	13Aa1	Teesdalio nudicaulis-Arnoseridetum minimae	12Aa5	Echinochloo-Setarietum
32	17Ac1	Agropyro reptentis-Aegopod. podagrariae	17Ab1	Alliario-Chaerophylletum temuli
33	12Ba1	Chenopodio-Utricetum urentis	12Aa3	Veronico-Lamietum hybridi
34	38Ab1	Stellario-Carpinetum	21Aa1	Koelerio-Gentianetum
35	12Bb1	Coronopo-Matricarietum	12Ba1	Chenopodio-Utricetum urentis
36	12Ba5	Melandrio-Berteroetum	12Aa5	Echinochloo-Setarietum
37	12Aa1	Chrysanthemo-Sperguletum	12Aa5	Echinochloo-Setarietum
38	13Ab1	Papaveretum argemones	17Ab1	Alliario-Chaerophylletum temuli
39	37Aa2	Fago-Quercetum	38Aa1	Carici remotae-Fraxinetum
40	30Ba1	Genisto pilosae-Callunetum	12Aa5	Echinochloo-Setarietum
41	17Aa2	Balloto-Chenopodietum	17Ab1	Alliario-Chaerophylletum temuli
42	38Aa7	Anthrisco-Fraxinetum	38Aa5	Violo odoratae-Ulmetum
43	12Bd1	Onopordetum acanthii	20Bc2	Tortulo-Phleetum arenarii
44	31Aa1	Trifolii (medii)-Agrimonietum	25Ba1	Arrhenatheretum elatoiris
45	38Aa3	Pruno-Fraxinetum	38Aa1	Carici remotae-Fraxinetum

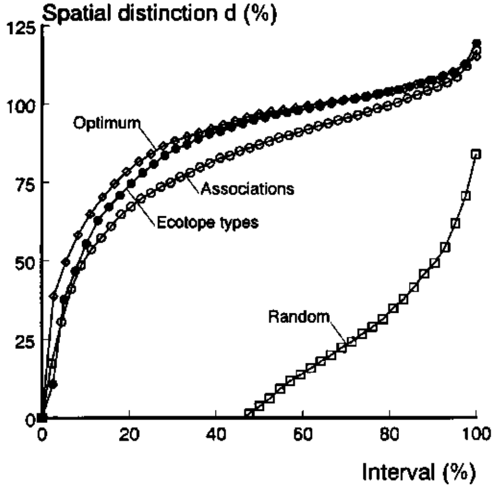

Figure 3.6: Spatial distinction *d* versus interval of similarity of both ecotope types and associations. Explanation: Fig. 3.4.

of associations is relatively small. Yet, from the results one might conclude that deriving detailed vegetation units is not without problems. The more we differentiate units the more difficult it becomes to find species whose distribution is confined to one unit. Consequently, the diagnostic value of the species diminishes. Not surprisingly, Shimwell (1971) argues that exclusive character species are rare at association level, whereas they are common at alliance level. The result is that the internal correlation does not increase from alliance to association but decreases instead.

3.5 Discussion

As is the case with most comparisons, those between ecological and phytosociological groups here presented are incomplete and imperfect. For example, I could only focus on the diagnostically most significant species (vascular plants) and I had to exclude trees and shrubs. The demand for a minimum of 3 diagnostic species per classification unit meant that not all units were taken up in the comparison. Yet, the samples created by this demand were distributed over the same types of environment. Besides, they were large enough for statistically highly significant statements about the differences between the two systems. I have not taken into account the fact that species in the Netherlands could reach the boundary of their area, and then only have a regional diagnostic significance. The km-squares on which the calculations are carried out are unevenly distributed over the Netherlands (Fig. 3.1). Finally, the large grid-cell level of 1 km^2 is responsible for spatial noise.

To my mind, however, the comparison is acceptable because these limitations are imposed on both classification systems. Moreover, there is no reason to think that the limitations of the comparison puts one classification system at a greater disadvantage than the other system.

This chapter dealt with the question whether the division of species into ecological groups (Runhaar *et al.*, 1987) is suitable for a FLORBASE-aided description of the vegetation of the Netherlands. If we take the phytosociological groups of Westhoff & Den Held (1969) for a standard, the answer should - in the light of the results presented here - be in the affirmative: *the division is certainly suitable.* In contrast to what might be expected the phytosociological species groups of Westhoff & Den Held did not appear to be better than the ecological species groups of the ecotope system. The internal correlations of the phytosociological groups are slightly higher, but the difference between the ecological groups is insignificant. Besides, the difference can partly be ascribed to the stricter selection of the character species. On the other hand, the ecological groups exhibit a significantly higher distinguishing capacity.

The hierarchy in Westhoff & Den Held's system is hardly reflected in the correlation calculations: spatially well correlated associations rarely come from the same alliances, and spatially well correlated alliances rarely belong to the same order. Conversely, spatially well correlated ecotope types are often taken from the same site type.

Whether or not the conclusions remain valid when the analysis is carried out in a different manner has been investigated in various ways. It has been tried, for example, to increase or decrease the minimum number of diagnostic species per unit, or to use the historic data from the ATLAS database (Section 1.4). However, the conclusions remain unaltered: there is no significant difference in internal correlation but there certainly is in distinguishing capacity.

To examine the descriptive capacity of the ecotope system, I also made comparisons with two other divisions: with the division of species into the site types of the NTM model (Section 2.2) and with the ecological species groups of Arnolds & Van der Meijden (1976). These two systems appear to be inferior to both the ecotope system, and Westhoff & Den Held's system: their internal correlations are lower and their external correlations higher.

Until now I have not dealt with the *causes* of the relatively good performance of the ecotope system. One ready explanation is that the phytosociological classification implicitly implies all sorts of factors which are not visible on the scale of a km-square. Consider for instance subtle differences in vegetation management or in micro climate. Apart from the classification characteristic 'vegetation structure', the ecotope system is based on factors with a clear geographical component at km-square level (nutrient availability, etc.). However, this explanation does not account for the slight differences in internal correlation: in disregard of the chosen division, species of the same unit should occur together spatially, even in grid-cells of 10 × 10 km or larger. The explanation only partly covers the difference in external correlation. If all kinds of non soil-bound factors are implied in Westhoff & Den Held's system then it must be at the expense of the 'soil factors'. If, for instance, the factor of micro climate is used as a classification characteristic, then - with a given number of classification units - less weight will be given to the factors salinity, moisture regime, nutrient availability and acidity. And these are exactly the factors known to be important to the spatial variation in the species composition of the plant cover.

Chapter 4

Ecosystem maps of the Netherlands

4.1 Introduction

The Dutch Government tries to have nature play a role in its policy. Therefore, it needs useful and clear information about the way nature manifests itself in the Netherlands. This chapter provides such information in the form of national distribution maps of ecosystem types, see the color maps of Figs. 4.26-4.53. It describes how these ecosystem maps are derived from FLORBASE-2c. In this publication, the ecosystem maps are needed for the geographical schematization for the DEMNAT-model (Chapter 2). Besides, they serve to determine the botanical valuation of kilometer squares (Chapter 5).

The ecosystem types of Figs. 4.26-4.53 have been derived from the ecotope system (Section 1.6), but for practical reasons, I do not use the system's original ecotope types. In the next section so-called *ecotope groups* are formed out of ecotope types. Additionally, a description will be given of the way indicator species have been ascribed to these new ecosystem units.

The data in FLORBASE-2c have been taken from different inventories (Section 1.4.2) and therefore the database is likely to contain *inventory effects*, i.e. geographical differences that have to be seen in relation with the way the inventories were made, rather than with botanical differences. Inventory effects are clearly visible on color Fig. 4.24, which shows the number of species per km-square in FLORBASE-2c. It appears that the rather uneven distribution of the numbers of species is mainly determined by the borderlines between the provinces (Fig. 4.1): especially the provinces of Noord- and Zuid-Holland, Zeeland, Drenthe and Utrecht have been investigated quite well, whereas few data are available from the provinces of Groningen, Friesland, Flevoland, Noord-Brabant and Limburg. However, regions with few data often contain many indicative species, and these are precisely the ones that are needed for the making of the ecosystem maps.

There are two reasons for the fact that less investigated km-squares contain relatively many indicative species. One reason is that botanists often especially

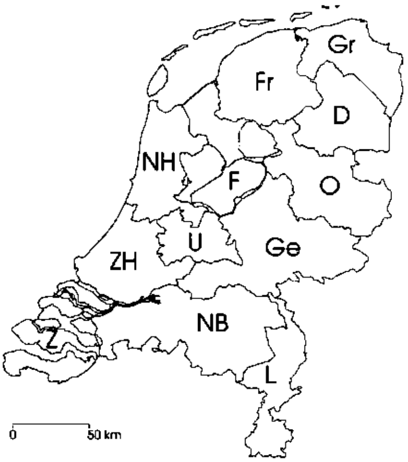

Figure 4.1: Division of the Netherlands in provinces: NB=Brabant, D=Drenthe, F=Flevoland, Fr=Friesland, Ge=Gelderland, Gr=Groningen, L=Limburg, NH=Noord-Holland, O=Overijssel, U=Utrecht, Z=Zeeland, ZH=Zuid-Holland.

recorded rare and indicative species, because they have a special interest in such species (Section 4.3.2). Another reason is that from the species that were only recorded on paper - and not stored in a digitally way - only the ecologically indicative ones have been digitalized for FLORBASE-2c.

To compensate for inventory-effects, a *gap-filling method* has been developed. This method - which will be described in Section 4.3 - results in a gap-filled version of FLORBASE-2c. Section 4.4 accounts for the way distribution maps of ecotope groups have been derived from this version. For the sake of convenience, these maps are called *ecotope maps*.

On the ecotope maps the degree of 'completeness' (botanical quality) has been indicated by a division in four classes: 'noise', 'low', 'high', 'very high' (Figs. 4.26-4.53). Such ordinal classes can hardly be of use for computation purposes. Therefore, Section 4.4.3 will deal with the possibility of expressing the concept 'completeness' in a number.

The reliability of the maps will be examined in Section 4.5 by comparing them with other data about the occurrence of ecotope groups. The boundaries of the completeness classes that are shown on the maps are based on expert judgement. In Section 4.6 it will be shown that this judgement was carried out in very systematic way. Subsequently, the possibility of imitating the expert judgement by formal calculation procedures will be looked into. Section 4.7 caps the chapter with an extensive discussion.

4.2 Composition of ecosystem types

4.2.1 Ecotope groups: aggregated ecotope types

In this chapter, nation-wide observations on the distribution of vascular plant species will serve to derive completeness figures of ecosystems per km-square. In order to be able to do this with reasonable reliability, it is necessary that every ecosystem type contains a sufficient number of indicator species. As this is not the case for many of the ecotope types, it is necessary to merge ecologically related types into aggregated types. For the sake of clarity, such aggregated types will from now on be called *ecotope groups*. For the aggregation the following rules are applied (Witte & Van der Meijden, 1993)[1]:

- Ecotope types of grasslands (G), pioneer vegetations (P) and tall herb vegetations (R) of one and the same site type are put together. As for the groups that are formed in this way, the classification characteristic 'vegetation structure' is indicated by a capital K (Dutch: 'Kruidvegetatie' = Herbaceous vegetation).
- Ecotope types of water vegetations (W) are joined with those of terrestrializing vegetations (V); the new structure code is indicated by a capital A (Aquatic). Additionally, ecotope types of waters with neutral acidity are joined with ecotope types of alkaline waters (code: A12).
- Ecotope types of woods and shrubs remain unchanged (structure code H, from the Dutch 'Houtige vegetatie' = Woody vegetation).
- The additional characteristics in the ecotope system are ignored, except for the characteristic 'calcareous' within the pioneer- and grassland vegetations of moist and moderately nutrient-rich sites. This ecotope group is indicated by the code 'K46' (herbaceous vegetations of moist, moderately nutrient-rich, calcareous soils).

These rules allow for the compilation of 38 ecotope groups (Tables 4.1 and 4.2). However, it would not be sensible to publish every one of the 38 distribution maps, since some of these maps are not reliable. In Table 4.1 these groups are indicated by the number 0.

The first ecotope groups to be left out are those of sites that are nutrient-rich as well as moist or dry (K47, K48, K67, K68, H48, H69), because their maps mainly show the influence of eutrophication. A large number of the locations where these ecotope groups occur at present were originally nutrient-poor for the precipitation excess of the Dutch climate washes away nutrients from moist and dry soils. Besides, these groups appeared to have been poorly investigated. Probably it often happened that florists did not deem it worthwhile to make sufficiently extensive inventories on these locations.

Other ecotope groups that have to be skipped are those with a very small number of indicator species (A11, H21, H41, H61) because a reliable determination of

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[1] This aggregation does not affect the conclusions that were drawn in Chapter 3: also for ecotope groups there is no significant difference in internal correlation with the phytosociological alliances and with the associations, but there is indeed in distinguishing capacity.

Table 4.1: The ecotope groups distinguished in the Netherlands. Example: ecotope group K21 consists of pioneer vegetations and grasslands on wet, nutrient-poor and acid soils (ecotope types P21 and G21, see Table 1.3). Explanation of the superscripts: 0 - neither selected for presentation on an ecosystem map, nor for the geographical schematization of DEMNAT; p - only selected for presentation; d - selected for presentation as well as DEMNAT.

	Fresh					Brackish	Saline
	Nutrient-poor			Moderately nutrient-rich	Very nutrient-rich		
	Acid	Neutral	Alkaline				
Water	A11^{0}	A12^{d}		A17^{d}	A18^{d}	bA10^{d}	
Wet	K21^{d}	K22^{d}	K23^{d}	K27^{d}	K28^{d}	bK20^{d}	zK20^{p}
	H21^{0}	H22^{d}		H27^{d}	H28^{d}		
Moist	K41^{d}	K42^{d}	K43^{p}	K46^{p}	K48^{0}	bK40^{d}	
	H41^{0}	H42^{d}	H43^{p}	K47^{0}	H48^{0}		
				H47^{d}			
Dry	K61^{p}	K62^{p}	K63^{p}	K67^{0}	K68^{0}	bK60^{p}	
	H61^{0}	H62^{p}	H63^{p}	H69^{0}			

completeness is not really possible for them.

Eighteen out of the 28 groups that have been selected for presentation are relevant for DEMNAT, since they are susceptible to changes in water management. In Tables 4.1 and 4.2 these groups have been indicated by the character 'd'. All 'dry' groups have been excluded from DEMNAT for the obvious reason that they are free from the influence of groundwater and surface water and, consequently, from that of water management. The same is true for a number of 'moist' groups (K43, H43, K46) that almost exclusively occur on soils with a very low groundwater table, but with enough available soil moisture. Ecotope group zK20 is irrelevant for DEMNAT, since it only occurs on grounds that are regularly flooded by seawater, which is not accounted for in this model.

4.2.2 Indicator values

In former times, the botanical quality of an ecotope group was derived from the *number* of indicator species that was found within one hour square (Witte & Van der Meijden, 1989; Witte & Van der Meijden, 1990) (Fig. 1.4). For FLORBASE a more subtle method has been developed, allowing for the ecological amplitudes of species (Witte & Van der Meijden, 1993; Witte & Van der Meijden, 1995). In this method species are given an *indicator value* v. This indicator value stands for the extent to which a species is characteristic for its ecotope group.

Table 4.2: Ecotope groups that are selected for presentation on a distribution map, together with the ecotope types (Table 1.3) they are derived from. The ecotope groups of DEMNAT-2.1 are marked with the character 'd'.

Ecot. group	Description	Ecotope types	Fig.
K21^d	Herbaceous vegetation on wet, nutrient-poor, acid soil	P21 G21	4.26
K22^d	Herbaceous veg. on wet, nutrient-poor, neutral soil	P22 G22	4.27
K23^d	Herbaceous veg. on wet, nutrient-poor, alkaline soil	P23 G23	4.28
K27^d	Herbaceous veg. on wet, moderately nutrient-rich soil	P27 G27 R27	4.29
K28^d	Herbaceous vegetation on wet, very nutrient-rich soil	P28 G28 R28	4.30
K41^d	Herbaceous vegetation on moist, nutrient-poor, acid soil	P41 G41	4.31
K42^d	Herbaceous veg. on moist, nutrient-poor, neutral soil	P42 G42	4.32
K43	Herbaceous veg. on moist, nutrient-poor, alkaline soil	P43 G43	4.33
K46	Herbaceous vegetation on moist, moderately nutrient-rich, limy soil	P47* G47* R47*	4.34
K61	Herbaceous vegetation on dry, nutrient-poor, acid soil	P61 G61	4.35
K62	Herbaceous veg. on dry, nutrient-poor, neutral soil	P62 G62	4.36
K63	Herbaceous veg. on dry, nutrient-poor, alkaline soil	P63 G63	4.37
H22^d	Woods and shrubs on wet, nutrient-poor, neutral soil	H22	4.38
H27^d	Woods and shrubs on wet, moderately nutrient-rich soil	H27	4.39
H28^d	Woods and shrubs on wet, very nutrient-rich soil	H28	4.40
H42^d	Woods and shrubs on moist, nutrient-poor, neutral soil	H42	4.41
H43	Woods and shrubs on moist, nutrient-poor, alkaline soil	H43	4.42
H47^d	Woods and shrubs on wet, moderately nutrient-rich soil	H47	4.43
H62	Woods and shrubs on dry, nutrient-poor, neutral soil	H62	4.44
H63	Woods and shrubs on dry, nutrient-poor, alkaline soil	H63	4.45
A12^d	Veg. in stagnant, nutrient-poor, neutral/alkaline water	V12 W12 W13	4.46
A17^d	Vegetation in stagnant, moderately nutrient-rich water	V17 W17	4.47
A18^d	Vegetation in stagnant, very nutrient-rich water	V18 W18	4.48
bK20^d	Herbaceous vegetation on wet, brackish soil	bP20 bG20 bR20	4.49
bK40^d	Herbaceous vegetation on moist, brackish soil	bP40 bG40 bR40	4.50
bK60	Herbaceous vegetation on dry, brackish soil	bP60	4.51
bA10^d	Vegetation in stagnant, brackish water	bV10 bW10	4.52
zK20	Herbaceous vegetation on wet, saline soil	zP20 zG20 zR20	4.53

*calcareous subtypes

Table 4.3: Indicator values v of species from ecotope group K21. Example: *Drosera rotundifolia* has been ascribed to ecotope types P21, G21 and G22; so its indicator value amounts to $\frac{2}{3}$ for ecotope group K21. Species numbers and names are according to Van der Meijden (1996).

55	Andromeda polifolia	1	858	Narthecium ossifragum	1
219	Carex curta	$\frac{1}{2}$	912	Oxycoccus macrocarpos	$\frac{1}{2}$
417	Drosera intermedia	1	913	Oxycoccus palustris	1
418	Drosera rotundifolia	$\frac{2}{3}$	962	Polygala serpyllifolia	$\frac{1}{2}$
1431	Erica scoparia	1	1068	Rhynchospora alba	1
473	Erica tetralix	$\frac{1}{2}$	1069	Rhynchospora fusca	1
476	Eriophorum angustifolium	$\frac{1}{3}$	1153	Trichophorum cespitosum	$\frac{1}{2}$
479	Eriophorum vaginatum	1	1330	Vaccinium uliginosum	1
568	Gentiana pneumonanthe	$\frac{1}{2}$	1394	Wahlenbergia hederacea	1
777	Lycopodiella inundata	1			

Table 4.3 lists - by way of example - a number of species that have an indicator value of at least $\frac{1}{3}$ for the ecotope group K21. Species with $v < \frac{1}{3}$ have not been considered as their indicative significance is negligible. In Table 4.3 *Andromeda polifolia* has an indicator value of 1 as this species is unique for K21, whereas *Carex curta* has a $v = \frac{1}{2}$, as there is another ecotope group (K22) to which it also belongs. However, *Drosera rotundifolia* has an indicator value of $\frac{2}{3}$ because this species belongs to two K21-ecotope types (P21 and G21), whereas it belongs to only one ecotope type (G22) from another ecotope group. Thus, v is calculated by dividing the number of ecotope types to which the species belongs within the ecotope group considered, by the total number of ecotope types to which the species belongs.

Among ecotope groups there is much variation in the number of indicator species, as well as in the spectrum of indicator values v (Table 4.4).

If the indicator values are added up for all m species in a km-square belonging to a certain ecotope group, the *indicator value score* S is obtained, being a combined measure for both the occurrence probability and the botanical quality of that particular ecotope group in the km-square:

$$S = \sum_{i=1}^{m} v_i \tag{4.1}$$

If, for example, in a certain km-square the species *Carex curta*, *Drosera intermedia*, *Eriophorum angustifolium* and *Rhynchospora fusca* occur, then S in that square amounts to $2\frac{5}{6}$ (Table 4.3: $\frac{1}{2} + 1 + \frac{1}{3} + 1$). The question how these scores should be used for construction of the maps will be answered in Section 4.4.1.

Table 4.4: Number of indicator species and composition of indicator values per ecotope group. Example: ecotope group K21 contains 19 indicator species of which 58% has an indicator value of 1.

Ecotope group	Number of species	Percentage of species with indicator value v of:			
		1	$\frac{1}{2}$	$\frac{1}{3}$	rest
K21	19	58	32	5	5
K22	67	36	40	24	0
K23	25	36	40	24	0
K27	88	28	42	26	5
K28	58	40	33	21	7
K41	12	17	67	17	0
K42	37	27	49	22	3
K43	59	42	39	17	2
K46	78	55	37	8	0
K61	12	17	33	50	0
K62	54	37	43	20	0
K63	100	47	36	16	1
H22	14	50	29	21	0
H27	30	10	60	30	0
H28	12	0	75	25	0
H42	41	39	46	15	0
H43	59	49	42	8	0
H47	45	31	47	22	0
H62	14	29	57	14	0
H63	16	44	44	13	0
A12	28	36	54	11	0
A17	61	36	49	15	0
A18	38	29	61	11	0
bK20	27	26	33	30	11
bK40	38	39	32	29	0
bK60	10	50	40	10	0
bA10	9	22	78	0	0
zK20	21	76	19	5	0

4.3 Gap-filling of flora databases

4.3.1 Method

In case a flora database contains large inventory effects, the results obtained from this database should be interpreted with considerable expertise. If one would refrain from doing so, then regions from which very few data are implied in a database might for instance turn out to be the best locations for groundwater extraction. Obviously this is an absurd conclusion, proving that flora databases should be submitted to extensive quality control and, if necessary: to *gap-filling.*

The gap-filling method here presented is based on the phenomenon that certain plant species may function as *guiding species* for other species: when these species are found it is very likely that other, often more common, species are also present. This method works particularly well if mainly the 'interesting' species were registered, whereas the common species with a broad ecological amplitude were left out (for instance because the presence of these species was taken for granted).

Species that, in a distributional sense, show a strong correlation with guiding species, are called *filling species*. A guiding species together with its associated filling species is called a *pair*. Pairs can be derived from the flora databases. For this purpose, it is calculated how - in a particular area, which is known to have been thoroughly examined beforehand - species overlap as to their distribution. To illustrate this, Table 4.5 gives part of a list with guiding and filling species. Behind each pair the so-called *overlap* O is mentioned, indicating the degree to which the guiding species overlaps its filling species in a distributional sense. The overlap between guiding species i and filling species j is calculated by dividing the number of cells in which both species are represented, by the total number of cells in which i occurs:

$$O_{i,j} = \frac{n_i \cap n_j}{n_i} \tag{4.2}$$

The overlap O indicates the probability that the filling-species is present, on the condition that the guiding species is present in the cell: it is a *conditional probability.*

A list of pairs of guiding- and filling species can be used for completing plant records, resulting in a new flora database, on the basis of which distribution maps can be made. To this end, it is necessary in the first place to determine above which value of O, species have to be completed. Only the pairs that meet this *overlap criterion* are selected. Subsequently, all cells in which the guiding species is registered whereas the filling species misses, may be completed with this filling species on the basis of those pairs. The pairs of Table 4.5 in combination with an overlap criterion of 0.90 will for instance lead to completing every grid-cell in which species 55 (*Andromeda polifolia*) occurs, with the species 473 (*Erica tetralix*) and 476 (*Eriophorum angustifolium*).

Table 4.5: Part of a gap-filling list. Only species of ecotope group K21 and overlap values O of more than 0.80 are shown. Example: in 89% of the grid-cells where species 55 is present, species 219 is present as well. Thus, the conditional occurrence probability of species 219 in a grid cell is 0.89, given the presence of species 55 in that cell.

Ecotope	Guiding species		Filling species		Overlap O
group	No.	Name	No.	Name	
K21	55	Andromeda polifolia	219	Carex curta	0.89
K21	55	Andromeda polifolia	417	Drosera intermedia	0.89
K21	55	Andromeda polifolia	418	Drosera rotundifolia	0.89
K21	55	Andromeda polifolia	473	Erica tetralix	1.00
K21	55	Andromeda polifolia	476	Eriophorum angustifolium	1.00
K21	219	Carex curta	473	Erica tetralix	0.90
K21	219	Carex curta	476	Eriophorum angustifolium	0.88
K21	417	Drosera intermedia	418	Drosera rotundifolia	0.86
K21	417	Drosera intermedia	473	Erica tetralix	0.98
K21	417	Drosera intermedia	476	Eriophorum angustifolium	0.92
K21	417	Drosera intermedia	568	Gentiana pneumonathe	0.85

It is inevitable that every form of gap-filling implies new errors, for instance because the relations that are found for a certain region are unsoundly extrapolated to other regions. In order to avoid these errors as much as possible the following rules are applied during the derivation of pairs of guiding- and filling species:

1. Only the overlap between species within one and the same ecotope group are examined.

   By examining exclusively the overlap between species of one and the same ecotope group apparent correlations, without any ecological significance, are avoided to the greatest possible extent. Table 4.5 for instance, only contains relations that have been determined between species of ecotope group K21.
2. The overlap should be statistically significant.

   Rare species will easily show a high overlap with very common species, without this result necessarily being statistically significant. Therefore, only species with a statistically significant overlap are selected (χ^2-test with $P < 0.01$; Parker, 1979).
3. Species, of which the borderline of their distribution area crosses the Netherlands are not used as filling species. (However, this does not alter their role as guiding species!).

   For a large number of species the borderline of their distribution area crosses the Netherlands, and it would not be sensible to 'complete' these species outside their area. Therefore, Witte & Van der Meijden (1993, 1995) proposed to make it a rule of thumb to exclude these species from completion, even if the completion should take place within their area. They selected such species from an article about the European and global distribution of the Dutch vascular species (Schaminée *et al.*, 1992), declaring those species of which the area was given the predicate 'marginal' or 'outpost', unfit for completion.

4.3.2 Validation of the gap-filling method

For all the demands that are made on the generation of a list of pairs, there is still the possibility that more errors are introduced than corrected. Therefore the question remains: *do distribution maps improve by gap-filling*? In order to deal with this question three tests have been developed by Witte & Van der Meijden (1993, 1995). FLORBASE had not yet been put into use at the time these test were developed, so the tests have been applied to the older ATLAS database, described in Section 1.4.2. An advantage of this database is that the quality of its inventories are well known. In the tests only those hour squares are used that, according to Van der Meijden *et al.* (1989), suit analytical purposes. Squares that largely consist of foreign country or water, as well as particularly insufficiently investigated hour squares have for instance not been taken into account. These are the three tests.

1. It is examined whether gaps are filled where we can expect them to be filled.
2. If gap-filling is allowed, then the number of gap-fillings in a certain area gives an indication about the quality of the inventories. Other indications for the quality of the inventories may be searched for; in case these coincide with the number of gap-fillings, probably the right areas have been chosen for gap-filling. It goes without saying that the number of gap-fillings can be no more than an *indicator* for the inventory quality. For instance, grid-cells lacking indicator species cannot be completed.
3. A set of records is randomly removed from the database, upon which it is tried to find out whether, through gap-filling, the characteristics of the original database may be regained.

Ad 1: Visual cheque locations gap-filling

The color map of Fig. 4.22 shows the results of gap-filling that is applied on the two inventory periods from the ATLAS database.

Towns with over 20,000 inhabitants according to Kwast (1934) are also shown in Fig. 4.22A. It is significant that from the 38 towns with more than 20,000 inhabitants, only 4 fall within the white area and grey area of Fig. 4.22A (white = 'poorly investigated', grey = 'high number of gap-fillings'), whereas these areas cover 55% of the total number of hour squares. All in all, it appears that the gap-filling in the period 1902 – 1950 is the *most extensive in the sparsely populated areas* of the Netherlands, and the *least extensive in and around the densely populated areas.* With some effort several towns and villages can be discerned on Fig. 4.22A: Leiden and Den-Haag, Amsterdam, Rotterdam, Dordrecht, Utrecht, Eindhoven, Nijmegen and Arnhem, Enschede, Zwolle, Groningen. Could it be that transport was such a problem, those days, that inventories were mainly made in and around one's hometown? It seems probable.

On comparing Fig. 4.22A with the distribution map, which Witte & Van der Meijden (1990) made of ecotope group A18 (terrestrializing and water vegetations of stagnant, fresh, very nutrient-rich waters) (color Fig. 4.23), we are confronted with a striking resemblance. It appears that the inventory effect manifests itself exceptionally clear in this group. The other maps (1902 – 1950) show this effect to a far lesser extent, if at all. One explanation is that *very nutrient-rich ecosystems have never been given special attention by florists*, as they were mainly interested in rare species. Another explanation is that florists disliked wet feet. The fact that there exist relatively few publications on former water vegetations is one indication for the latter explanation. Another indication can be found in the large picture library of the National State Herbarium of the Netherlands. On the photographs from the first half of this century we mainly see florists (often gentlemen wearing a suit and a top hat), safely inspecting the land (Fig. 4.2), whereas florists who venture out into the water are seldom seen. The reason for this might be that *proper boots were scarce and expensive those days.* According to W. Blok (personal communication) - director of the Dutch museum for footgear, leather and leatherware in Waalwijk - it was not until the fifties that affordable, mass-produced synthetic boots were available on the market. Before that time there was nothing but expensive, hand-made footwear available (Fig. 4.3). This explains why we never see florists wearing boots on old fieldtrip photo's. However, occasionally we do see them with bandages swathed around their legs (Fig. 4.4). It goes without saying that these bandages are not suitable to stand in the water with for long, or to venture out on soggy terrain with. The absence of adequate footwear is likely to be the explanation for much of the 'progress', pictured by the distribution map of ecotope group A18 (Fig. 4.23): the progress is probably largely an artefact (compare especially Figs. 4.22A and 4.23A).

As to the period 1950 – 1980 (Fig. 4.22B) gap-filling is especially *low in the province of Noord-Brabant.* The high completeness of this province's data originates mainly from the thorough investigations of one florist: Mr J.M.A. Cools (1989). Furthermore, Fig. 4.22B shows that the distributional differences between the number of gap-fillings coincide, to some extent, with the *borderlines of the provinces* (which are plotted in this figure). This may be explained by the fact that part of the provincial inventory results have been incorporated in the ATLAS database (see Section 1.4.2).

Ad 2. Similarity with other indications for the quality of the inventory

An indication for the quality of the inventory is the number of common species per grid-cell. The reason for this is that florists are mainly interested in species that are relatively rare, so that they only register common species when a special request has been made to do so. This is for instance the case with the systematic inventories, such as have been carried out by several provinces or flora study groups. Fig. 4.5A shows the relation between the number of gap-fillings and the number of common species. Common species are defined as species that in the 1990-Standard list (Van der Meijden *et al.*, 1991) are classified in the Hour square Frequency Class UFK (Section 1.4.4) of 7, 8 or 9. From Fig. 4.5A it becomes

Figure 4.2: Floristic excursion near Roden, 1923. Should these gentlemen venture out into the pond behind them, they would get wet feet!

Figure 4.3: A leather boot from ca. 1910 (collection Museum of footgear, leather and leatherware in Waalwijk). The sole is nailed to the upper part with wooden pegs that swell in contact with water, making the boot almost waterproof. According to W. Blok (personal communication) one pair of these boots must have cost a two-and-a-half month wage of a workman. In current prices, this is at least 2,000 Dutch guilders (ca. 1,000 U.S. dollars).

Figure 4.4: Floristic excursion during rest near St. Geertuid, 1924. On the left, two participants with bandages swathed around their legs.

clear that the two inventory-indications show much resemblance: *the smaller the number of gap-fillings, the larger the number of common species.* Between the number of gap-fillings and the number of rare species there appears to be no relation at all (Fig. 4.5B).

Against the above result it could be objected that the determined relation is artificial, as the filling species is nearly always more common than its guiding species, which means that relatively many common species are used as filling species. In other words: there is hardly any gap-filling to be done in cells where many common species occur. To determine whether or not this objection is justified, also the number of gap-fillings has been plotted against the number of common species, to the exclusion of the species that are chosen for gap-filling. In this way, the relation becomes less clear ($r_s = -0.71$ instead of $r_s = -0.82$) but nevertheless the two indications continue to show resemblance.

When a certain area, such as a province or a flora district, has been thoroughly examined, many species may be expected to occur there that are highly correlated in a distributional sense. It is not that highly correlated pairs do not occur in poorly examined areas, but, due to the inadequate inventory, the data fail to express this correlation. Hence, the number of highly correlated pairs within an area gives information about the reliability of the inventory. Fig 4.6 shows the average number of gap-fillings per hour square plotted against the number of highly correlated pairs for each province (see Fig. 4.1 for the province codes that have been used in this figure). The two indications correlate: *the larger the number of gap-fillings, the smaller the number of highly correlated pairs.*

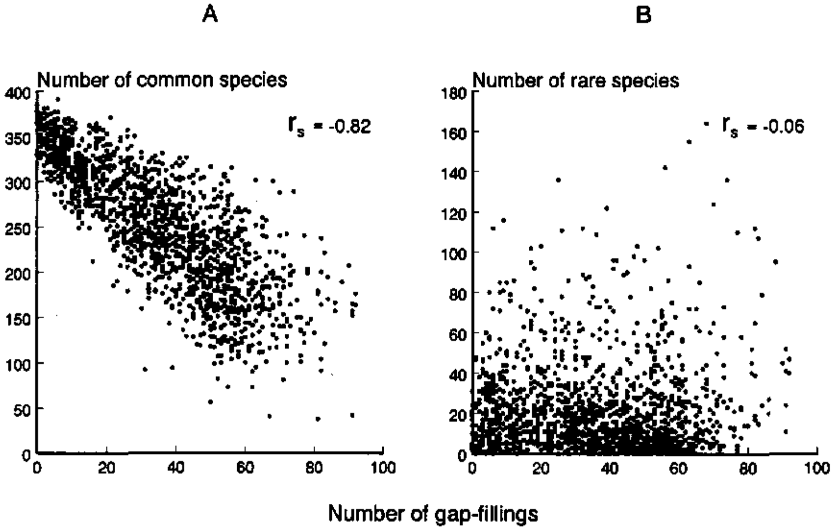

Figure 4.5: Relation between the number of gap-filled species and: A - the number of common species ($UFK > 6$); B - the number of rare species ($UFK < 6$) per hour square (ATLAS database 1950 – 1980). For technical information, see Witte & Van der Meijden (1993). r_s = Spearman rank correlation coefficient (with ties).

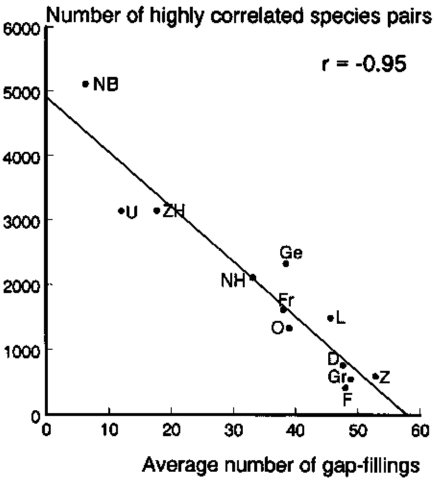

Figure 4.6: Relation between the average number of gap-filled species per hour square and the number of highly correlated species pairs per province (ATLAS database 1950–1980). The codes refer to the provinces, see Fig. 4.1. For technical information, see Witte & Van der Meijden (1993).

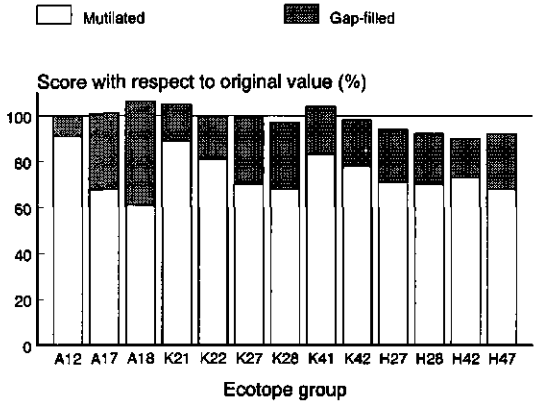

Figure 4.7: Mean scores of various ecotope groups, before and after gap-filling of a mutilated database (after: Witte & Van der Meijden, 1993).

Ad 3. Gap-filling of an artificially mutilated database

The most convincing test by Witte & Van der Meijden (1993, 1995), is the one where species are randomly removed from a database. After gap-filling, it appears that the characteristics of the original database have returned.

Fig. 4.7 shows how the average indicator value scores S (Section 4.2.2) of a number of ecotope groups have changed by randomly removing species from the ATLAS database (1950 – 1980). The decrease of the average scores ranges from 9% to 40% (respectively A12 and A18). It also shows how the average scores have increased by gap-filling. For all ecotope groups, gap-filling results in scores that are closer to the original values than the mutilated scores. The deviation from the original values ranges from the extremes -9 to 6% (respectively H42 and A18).

Fig. 4.7 does not yet show whether the right grid-cells have been filled up. To find out about this the gap-filling per ecotope group may be judged by plotting the original scores against the mutilated and gap-filled scores respectively. After a perfect gap-filling all points should be situated on the 1 : 1-line through the origin. Fig. 4.8 shows the result for ecotope group K28, making it clear that gap-filling is a sound method. Not only do the points approach the 1 : 1-line, they also get less scattered. This is also expressed by the figures for the Root-Mean-Square (RMS), plotted in Fig. 4.8: $RMS = 6.69$ for the mutilated score and $RMS = 1.16$ for the gap-filled score[2].

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[2] RMS is a general measure for the deviation of observed values with values that are calulated with a given function or a given model:

$$RMS = \sqrt{\frac{1}{n}\sum_{i=1}^{n}(y_{\mathrm{obs}i} - y_{\mathrm{cal}i})^2} \tag{4.3}$$

Where: n = number of observations, y_{obs} = observed value, y_{cal} = calculated value. The more RMS approaches 0, the more the function approaches the observed values.

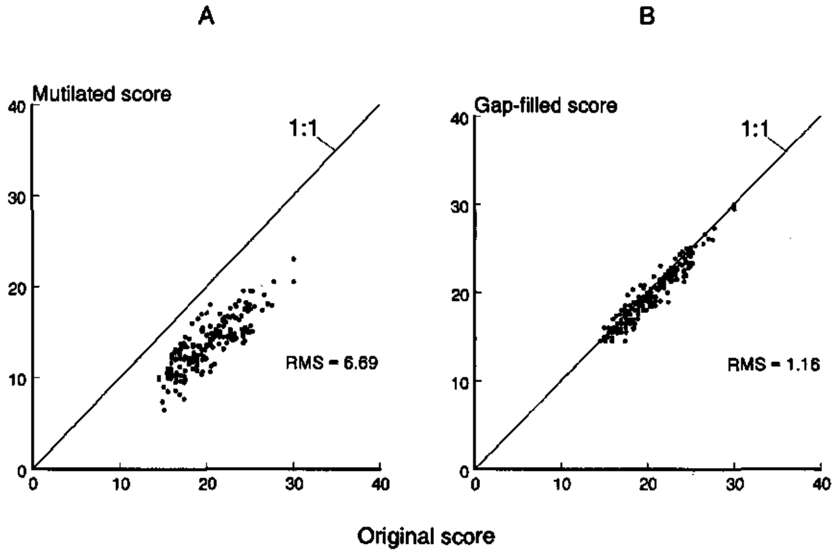


Figure 4.8: Relation between original scores of ecotope group K28 and: A - mutilated scores; B - gap-filled scores (after: Witte & Van der Meijden, 1993). After gap-filling, the scores more resemble the original values.

Conclusion

All three tests to which the gap-filling method has been submitted firmly support the conclusion that it is a *sound method for improving distribution maps.*

4.3.3 Gap-filling of FLORBASE

Method

For FLORBASE-2c the extent to which gap-filling takes place is made dependent on the number of common species per km-square. As was shown in the previous section, the number of common species gives information on the quality of the inventory. Generally speaking, it may be stated: the lower this number, the more gaps have to be filled up.

First, the km-squares have been classified in the classes of Table 4.6 according to their number of common species ($UFK > 6$). Then the overlap criterion is determined for each class, allowing optimal gap-filling averagely per class. This is facilitated by the use of pairs of guiding- and filling-species that have been derived from well-examined km-squares, here defined as km-squares that harbor at least 90 common species. The reason for choosing this minimum number is that even in areas that are poor in species by nature, at least 90 common species can be found, provided that the areas are very well investigated.

To enable judgement as to whether a class has been optimally completed a *mea-*

Table 4.6: Classification of km-squares on the basis of the number of common species.

Class k	Number of common species	Average number	Number of km-squares FLORBASE-2c
1	0	0.0	589
2	1 - 5	3.0	2480
3	6 - 10	8.0	1297
4	11 - 20	15.5	1580
5	21 - 30	25.5	1079
6	31 - 50	40.5	1933
7	51 - 70	60.5	1941
8	71 - 90	80.5	2597
9	91 - 110	100.5	3122
10	111 - 150	130.5	6573
11	> 150	-	7621

sure is needed for the inventory's quality, as well as a *standard*, allowing this measure to be tested. From a practical point of view, a simple measure has been chosen, coming down to *the number of common indicator species* in the km-square concerned, α. Consequently, these are the species that have an indicator value of $\frac{1}{3}$ or more for at least one of the ecotope groups, as well as a $UFK > 6$. The average α of class 10 from Table 4.6 has been chosen as a standard, $\overline{\alpha}(10)$. So the optimal overlap criterion per class k is determined in such a way that, after gap-filling, $\overline{\alpha}(k)$ amounts to $\overline{\alpha}(10)$.

In Fig. 4.9 the avarage number of common species per class k (Table 4.6) has been plotted against the overlap criteria that were obtained in this way. A regression line is drawn through the dots. Because of the fact that class 10 served as reference and that there was no need for class 11 to be filled up, these classes have not been implied in Fig. 4.9. Similarly, the two lowest classes (0 and $1 - 5$ common species) have not been considered, since the overlap criterion that satisfies the standard is unacceptably low (so the two points that are not shown are far below the regression line).

The regression line of Fig. 4.9 serves to determine a gap-filling criterion for every km-square by way of the number of common species. Subsequently, these criteria have been used to compile a gap-filled database, serving the drawing of distribution maps.

Results

Color Fig. 4.25 shows for every km-square the number of species that were added to FLORBASE-2c by means of gap-filling. As was to be expected, the gap-filling is highest when special efforts have been taken to collect or digitize data about indicative species: e.g. in the provinces of Noord-Brabant and Limburg, and in the north-western part of the province of Groningen. In case of few gap-fillings or

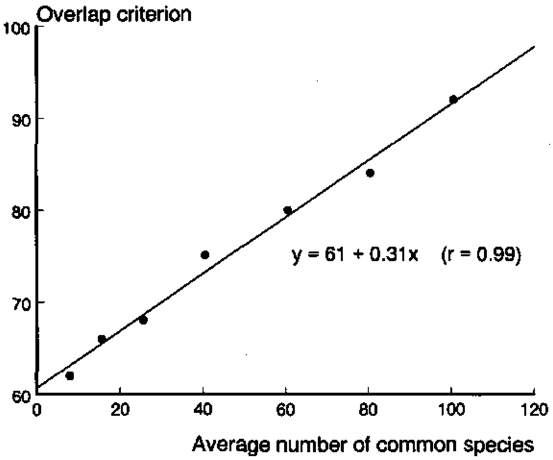


Figure 4.9: Relation between the number of common species per km-square and the optimized overlap criterion.

non at all, there may be two reasons for this: either the area has been examined especially well, so that there is no need for gap-filling, or it has been examined very poorly, so that gap-filling cannot take place for lack of guiding species.

Fig. 4.10 shows the average number of species that have been filled up, depending on the number of common species in a km-square. The gap-filling decreases, as the number of common species increases, down to a number of 130 common species per km-square where gap-filling is no longer needed. However, up to a level of ca. 20 common species the diagram shows a significant increase. An explanation for this is that squares that harbor very few common species contain merely separate records. According to this explanation, systematically examined squares would harbor more common species, together with the guiding species that are used for gap-filling.

Fig. 4.11 shows the increase of the average score per ecotope group in terms of percentage. Especially nutrient-rich groups appear to have been completed.

Decisive for the success of the gap-filling is (of course) its effect on the ecotope maps. Gap-filling has resulted in better maps: the calculated distribution pattern now relates much better than before to the expectations of experts in the botanical field (Witte & Van der Meijden, 1993). Gap-filling in well-examined provinces does not affect the picture. For comparison, also two distribution maps without gap-filling are presented (Ecotope groups K21 and A18, Figs. 4.54 and 4.55).

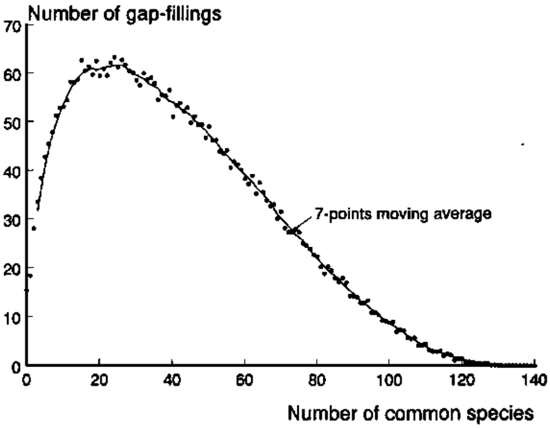


Figure 4.10: Relation between the number of common species and the number of gap-fillings per km-square.

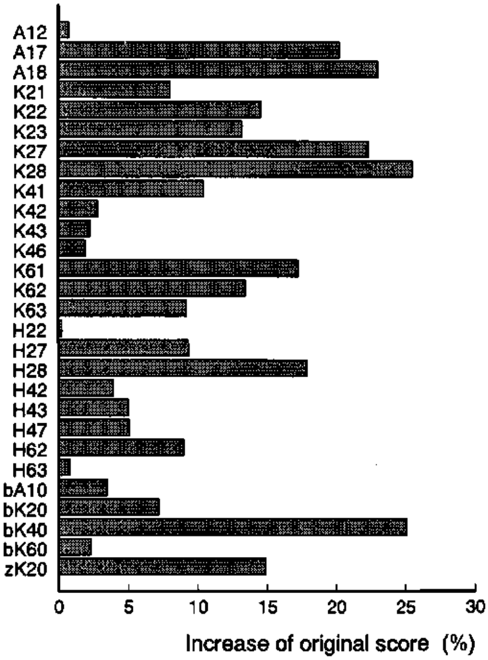


Figure 4.11: Increase in the average scores of ecotope groups as a result of gap-filling.

4.4 Distribution maps

4.4.1 Threshold values T and completeness classes

From the gap-filled FLORBASE-2c the indicator value score S per km-square (Section 4.2.2) has been calculated for every ecotope group. No direct use can be made of this score for the distribution maps because the ecotope groups differ in the number of ascribed indicator species, as well as in the spectrum of indicator values (Table 4.4). Besides, there are differences as to the percentage of indicator species that may actually be expected to occur within a km-square. Therefore, the indicator value score may very well indicate botanical quality differences *within* an ecotope group, whereas a comparison *between* the ecotope groups cannot directly be made. Following Van der Meijden (1977), Witte & Van der Meijden (1993, 1995) solved this problem by using *threshold values* T for every ecotope group, allowing scores to be classified in *completeness classes.* The notion *completeness* indicates the variable that, just like the score, provides information about the ecotope group's botanical quality, allowing, unlike the score, the groups to be directly compared with another.

The *lowest threshold value* (T_1) determines whether, according to the floristic information, an ecotope group may be said to be really present in a km-square, instead of classifying its occurrence as *noise.* The threshold values T_2 and T_3 divide the km-squares that harbor an ecotope group in three classes: *low*, *high* and *very high.* Table 4.7 provides for every ecotope group the values that have been determined for T_1 and T_3. As the second threshold value T_2 is situated right in between T_1 and T_3, there was no need to imply it in the table.

The threshold values of Table 4.7 are based on *expert judgement* (by R. van der Meijden of the National State Herbarium). A three-step-procedure was followed for their determination. First T_1 was determined by estimating the minimal score above which it is allowed to assume the ecotope group to occur with some botanical quality in the km-square. Subsequently, T_3 was chosen as the score, which may justifiably be qualified as 'very high'. And finally, T_2 was determined as the value located right in between T_1 and T_3.

Table 4.7: The first and third treshold values (T_1 and T_3) of the ecotope groups, meant for the gap-filled data of FLORBASE (after: Witte & Van der Meijden, 1995).

Ecot. group	T_1	T_3	Ecot. group	T_1	T_3	Ecot. group	T_1	T_3	Ecot. group	T_1	T_3
K21	2.0	7.0	K43	2.5	4.7	H28	1.7	2.3	A17	12.0	18.0
K22	4.0	9.0	K46	6.0	9.0	H42	4.0	6.0	A18	8.0	16.0
K23	2.5	4.5	K61	2.0	3.8	H43	5.0	9.0	bK20	3.4	6.2
K27	12.0	19.0	K62	5.8	10.1	H47	6.5	10.5	bK40	5.0	7.0
K28	10.0	19.0	K63	13.0	22.0	H62	1.4	3.0	bK60	2.0	3.8
K41	2.2	4.2	H22	2.2	3.2	H63	3.0	5.6	bA10	2.0	3.0
K42	2.5	4.9	H27	3.5	5.5	A12	2.2	5.0	zK20	5.2	9.2

With the above temporary threshold values, distribution maps were drawn. In case these maps were not immediately satisfactory the threshold values were adapted, taking into account aspects like the presence and the absence of ecotope groups in well-known areas.

The calculation of S here presented has been determined after studying various calculation methods. Consideration has been given to the possibility of giving rare species a higher indicator value v, based on the assumption that rare species have a relatively narrow ecological amplitude, causing them to be better indicators. Indeed it appears that the higher the indicator value score S, the larger the contribution rare species pay to that score (Fig. 4.12). However, according to Witte & Van der Meijden (1993) adding extra weight does not improve the maps. Quite the contrary: it makes them worse. This may be caused by the fact that rarity is already implied in v: rarity and v are correlated in a positive sense (Fig. 4.13).

Witte & Van der Meijden (1993) have paid ample attention to the question how high T_1 should be. T_1 determines whether or not a certain ecotope group is present. In other words, it determines whether or not there is a vegetation (plant community) that can best be characterized by the ecotope group in question. Especially if several species from one ecological group are found together, the presence of such a vegetation stands a fair chance. Another condition for the determination of T_1 is that exclusively vegetations should be detected that possess a certain botanical quality. For example, the observation of only *Lemma gibba* (UK: Fat duckweed; NL: Bultkroos) does not justify the assumption of the presence of ecotope group A18, although *Lemma gibba* requires very nutrient-rich water, even if it were but in one polluted puddle within a km-square. So for all ecotope groups a first threshold value has been determined, to the extent that it exceeds the score that one or a few species would have yielded. As the low score - called noise - is ignored, *relatively poor vegetations are not considered.*

4.4.2 Results: distribution maps

Fig. 4.14 gives a selection of km-squares that are considered reliable enough for the derivation of the completeness of the ecotope groups. For this, after gap-filling at least 50 plant species have to be present in a km-square. It should be noticed that this limit is rather arbitrary, and that its results in the omission of only those km-squares that are poorly investigated.

The gap-filled distribution maps are shown on the color maps of Figs. 4.26-4.53. The accompanying captions (modified after Witte & Van der Meijden, 1995) often contain information about the map's reliability, as well as about the ecotope group's area of distribution. The plant-geographical districts that are mentioned in some of the captions comply to the classification by Weeda (1990). The four colors that have been used for the maps have the following meaning:

Grey	The group is poorly developed or absent (completeness class 'noise'), or there is a lack of data (Fig. 4.14)

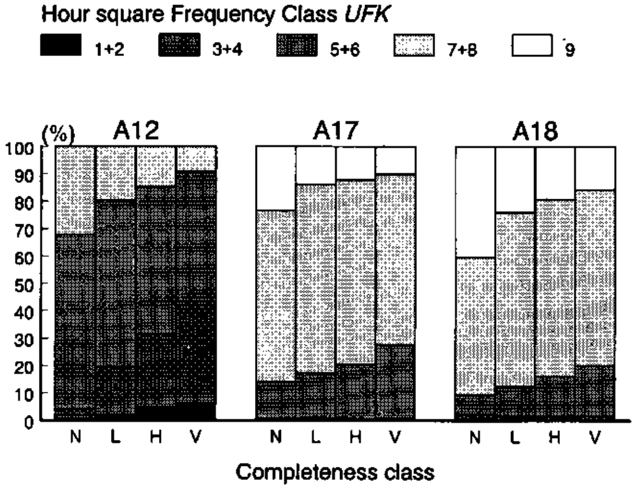


Figure 4.12: Average composition of the *UFK* in the scores of ecotope groups A12, A17 and A18, itemized per completeness class: N - noise, L - low, H - high, V - very high.

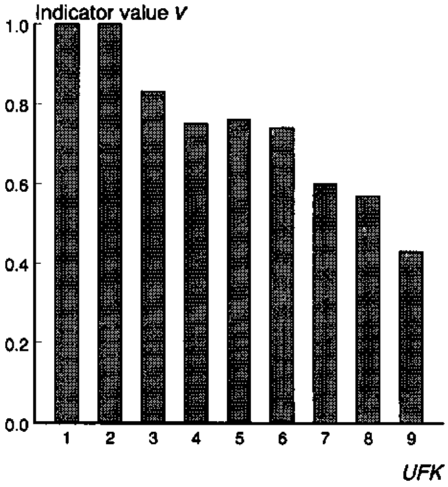


Figure 4.13: Average indicator value per Hour square Frequency Class *UFK* of ecotope group A17.

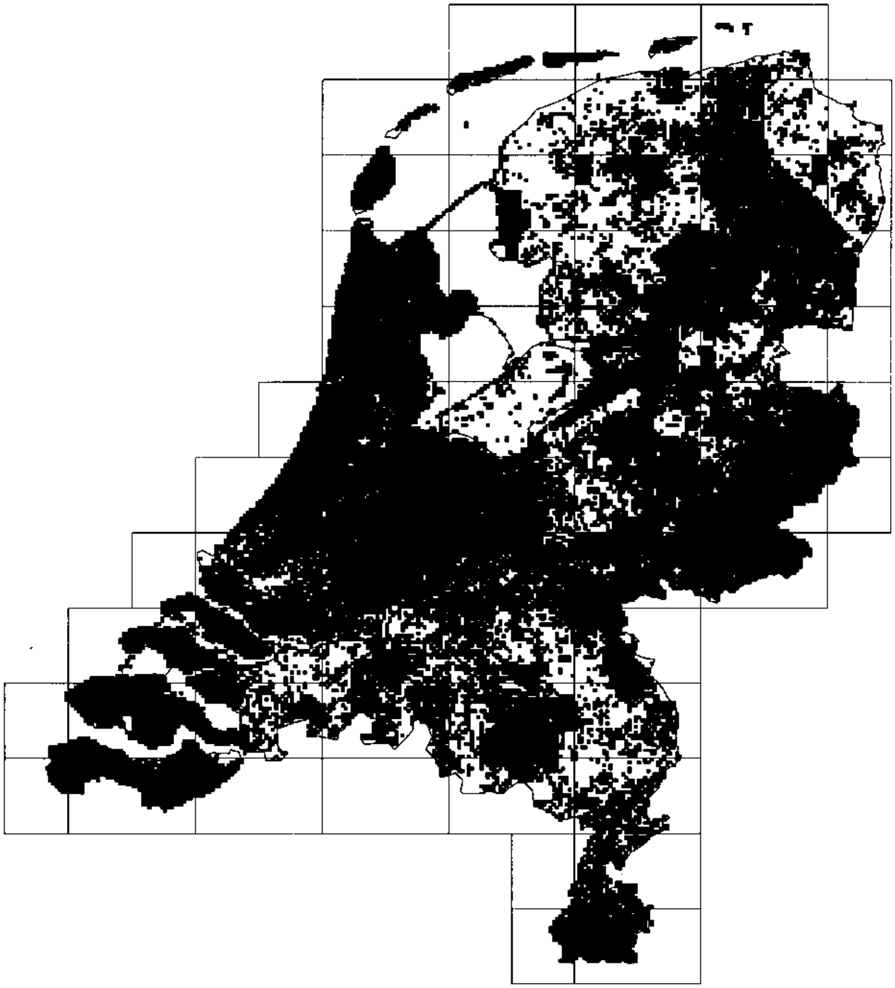


Figure 4.14: Km-squares with more than 50 species after gap-filling (FLORBASE-2c). It is assumed that these cells are reliable enough for presentation on the ecotope maps as well as for the geographical schematization of DEMNAT-2.1.

Blue	Ecotope group is probably present, but only moderately developed (completeness class 'low')
Yellow	Ecotope group is almost certainly present and also well-developed (completeness class 'high')
Red	Ecotope group is present beyond doubt and also very well-developed (completeness class 'very high')

So to facilitate presentation, the km-squares that are classified as 'noise' as well as the squares about which insufficient floristic information is available (according to Fig. 4.14) have been given the same color (grey). For all maps it is noted that data concerning the provinces of Friesland, Groningen (most of it) and Flevoland may be incomplete to a large extent. The same should be said about parts of the provinces of Overijssel, Gelderland, Noord-Brabant and Limburg.

4.4.3 The completeness fraction C

For calculations, such as are carried out by DEMNAT (Chapter 2) and by quantitative conservation value methods (Chapter 5), the qualitative notions 'noise', 'low', 'high' and 'very high' are not enough. Therefore, the completeness is here expressed in a *completeness fraction* C, which is a function of the indicator value score S. Witte & Van der Meijden (1993) approach this function as follows (Fig. 4.15):

$$C = \begin{cases} 0 & \text{if} \quad S < T_1 \\ \dfrac{S - T_1}{T_3 - T_1} & \text{if} \quad T_1 \le S < T_3 \\ 1 & \text{if} \quad S \ge T_3 \end{cases} \tag{4.4}$$

where:

C = completeness fraction
S = indicator value score
T_1 = first threshold value
T_3 = third threshold value

Scores within the noise class are given a completeness fraction of 0, as it is doubtful whether the ecotope group in question is present. Within the completeness class 'very high' the ecotope group is saturated, as it were, with characteristic species, so that all scores in this class are given the value $C = 1$. Scores from the completeness classes 'low' and 'high' are linearly interpolated between T_1 and T_3, as is shown in Fig. 4.15.

In practice, it appears that the relation between S and C, as shown in Fig. 4.15, yields very satisfactory results when the ecotope maps are combined into one conservation value map (Chapter 5). De Baere *et al.* (1986) proposed a similar relation, which was based on experiences with the valuation of grid-cells. For

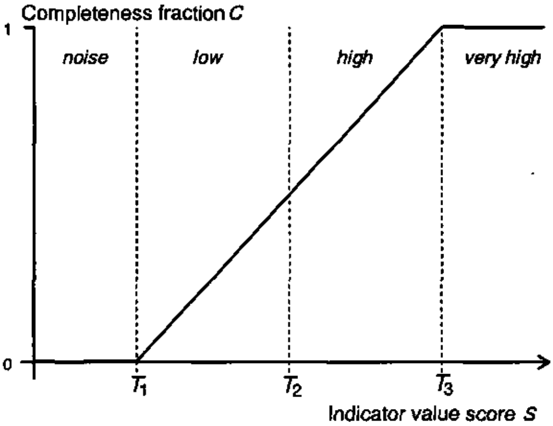


Figure 4.15: Completeness fraction versus the indicator value score. Also the completeness classes and the threshold values T are shown.

the province of Vlaanderen (Belgium), these authors combined 12 maps with the botanical quality of 'socio-ecological' species groups on a grid of 4 × 4 km-cells (Belgian hour squares).

It may seem surprising that, within km-squares that are qualified as 'very high', actual differences in the total number of characteristic species do not lead to further differentiation of the judgement. Firstly, this is due to a national quality scale: regional differences in the species richness within a very well-developed ecotope group should *not* lead to differences in completeness. Such differences are for instance caused by differences in the distribution areas of species, and to a great extent the latter are related to differences in the speed with which species invaded the Netherlands after the latest ice-age. Secondly, the differences in the total number of species within a fully developed ecotope group are mainly caused by the differences in the number of rare species. The majority of species that is rare on a national level is also rare within the ecotope group to which it belongs. The occurrence of such rare species is dependent on the variations, within the ecotope group, that are not covered by the classification characteristics of the ecotope system. Little is known about the reasons why some rare species, considered characteristic for a certain ecosystem, are sometimes absent ("the occurrence depends on chance"). Besides, it appears that rare species rarely occur in fixed combinations: the rare species in one place often differ from those in another (R. van der Meijden, personal communication). Yet, all those rare species are said to belong to one and the same ecotope group. In practice, this comes down to the situation that substantial differences in number and combination of species may occur, no longer relating to the differences in C.

As for Western Europe, it is clear that there is a trend-like increase in the total number of plant species from North to South. Besides, it may be concluded that the number of species per grid-cell in the Atlantic region is smaller than that in the (sub)continental region. Within a small country such as the Netherlands, actual climatic differences do not yet play an important role, but there is no doubt that they influence the number of species that can be expected within a group. In case the method would be applied to a much larger area than that of the Netherlands, ecotope groups would have to be divided in climatic variants, each with its own completeness classification.

4.5 Validation of the ecosystem maps

4.5.1 Introduction

Testing of the ecotope maps here presented can only take place to a limited extent, as no comparable nation-wide maps are available. The lack of such maps, after all, was the very reason for developing them. What we can do, however, is examining whether vegetation relevés with the threshold value method get allotted to the same ecotope group as is the case with a method especially developed for relevés (Section 4.5.2). Furthermore, a comparison can be made with nation-wide maps of a different kind (Section 4.5.3).

However, while making the two above comparisons, the following should be considered. In nature, there are usually no sharp borderlines. Especially in species-rich situations a clear borderline is exceptional, as vegetation gradually changes from one type into another. Where exactly the line between two types should be drawn, depends on the mapper's judgement as well as on the purpose, for which the map is made. Consequently, different mapping methods may lead to different maps. This becomes a problem as soon as dissimilarities are discovered when comparing one map to another. Should one method be qualified as worse than the other? Or do the two maps show something different and, if so, what constitutes this difference? Equally, when comparing ecotope maps with maps of a different kind, we have to take into account that the two maps were developed along different lines.

Another limitation is the fact that the only relevant criterion for a test is whether or not a certain ecotope group is present. The botanical quality of the groups cannot be tested for a lack of comparison material. On the other hand, we do have means for examining the extent to which the expert judgement was systematic. A test for this will be expounded in Section 4.6.

4.5.2 Comparison with vegetation relevés

Van Schadewijk (1993) compared the threshold value method with a method - incorporated in the computer program ECOTYP (Groen *et al.*, 1991) - which was especially developed for vegetation relevés. A random sample of 986 vegetation relevés was classified both with the threshold value method and with ECOTYP, upon which the results were compared.

With ECOTYP, 666 out of 986 relevés were assigned to one of the 28 ecotope groups presented in this chapter. With the threshold value method, 392 ecotope groups were found, 355 of which coincided with the classification according to ECOTYP. *So in 91% of the cases, the threshold value method puts relevés in the same ecotope group as ECOTYP.* This is a strikingly high percentage, all the more in view of the fact that merits of ECOTYP are not indisputable either. There is no doubt that there are quite a few relevés that the threshold value method failed to register, a consequence of the fact that, in the case of species-poor relevés, the indicator value score S did not exceed the first threshold value T_1.

Other reasons why the test cannot be fully relied upon are the fact that certain ecotope groups are not fully represented in the data Van Schadewijk investigated, as well as the fact that the threshold values have been determined for *gap-filled kilometer square data* instead of for vegetation relevés.

4.5.3 Comparison with other maps

A test, in which maps would be compared with other nation-wide data, would probably be preferable. However, as to botanical values in the Netherlands, maps that are directly comparable are not available.

The Nature Value Map 1988 (Bakker *et al.*, 1989) may serve this purpose, but only to a very limited extent, since it contains rough mapping units, such as 'woods' and 'bogs'. The only possibility is joining the mapping units 'dry or wet heath' with 'bogs' and comparing them to the combination K21, K41 and K61. The conclusion to be drawn from this is that the two maps overlap for about 50%. No bad result, considering that the data for the Nature Value Map were supplied by various experts from various provinces, and that the map does neither distinguish heath smaller than 5 ha, nor heath with over 20% shrubs and trees. In this context it is worthwhile mentioning that most of the Dutch heath terrains are smaller than 10 ha (Werkgroep Heidebehoud en Heidebeheer, 1988).

Equally, the maps of some ecotope groups have been compared by Witte *et al.* (1995) with the ecotope maps of the project 'Landscape Ecological Mapping of the Netherlands' (Landschapsecologische Kartering Nederland: LKN). These LKN-maps show the distribution of ecotope groups in km-squares for about 40% of the Netherlands. They contain information about the surface area of ecotope groups per km-square, based on data that were collected in different ways for each province and that, as a consequence, were transformed to the various ecotope groups with different translation procedures (Van der Linden *et al.*, 1995). This explains why the LKN-maps show all kinds of provincial differences (artifacts) that render the maps less suitable for analysis on a national scale. Understanding the comparison's results demands too extensive an explanation, so I refrain from evaluating it. However, some of its findings and conclusions are worth noting.

In all cases, the comparison reveals that there is significant, if not strong, resemblance between the two types of maps. The resemblance increases, as the ecotope group's completeness rises. However, it should be noted that the maps in some cases differ considerably. To some extent, this is caused by the fact that the maps were developed for different purposes: with the threshold value method, it was tried to indicate *botanical quality*, whereas with the LKN-method *average site circumstances* were mapped as adequately as possible, irrespective of the botanical quality. This difference explains for example why the LKN-method indicates more surface for the nutrient-rich ecotope groups: in quantitative respect these groups are well represented, though the botanical quality is often of minor interest. Ecotope group A18, for instance, occurs according to the LKN-method, in practically all clay- and fen-areas, even in km-squares with a very species-poor aquatic vegetation. The km-squares with A18 according to the threshold value method (Map 4.48) are for 96% within the area that is indicated by the LKN-method, but according to the threshold value method the ecotope group's distribution area is half the size!

Witte *et al.* (1995) also compared various ecotope maps with the site maps of Klijn *et al.* (1996, 1997). The site types of these maps were defined according to the same operational site-classes as those of the ecotope groups (see Section 2.3.4), enabling a direct comparison. Another advantage of these maps is that - being derived from one database (i.e. the $1:50,000$ soil map of the Netherlands), which was always interpreted according to the same systematics - they hardly suffer from artifacts. A considerable drawback is, however, that they register the *potential* occurrence of site types, instead of the actual occurrence, as ecotope maps do. As a result from eutrophication, for instance, the distribution area of nutrient-poor ecotope groups may expected to be smaller than that of the corresponding potential site types. However, eutrophication may also lead to the occurrence of nutrient-rich groups on soils that are originally nutrient-poor (e.g. A18 in the Pleistocene Netherlands, see Map 4.48).

Table 4.8 lists the overlap of the various ecotope groups with their site types for every completeness class. It shows that, generally spoken, *the ecotope maps correspond very well with the site maps.* This means that the occurrence of ecotope groups largely ranges within the area indicated on the site maps and that this is a very significant result ($p < 0.001$). *The correspondence increases with the completeness class.* In the following three cases, the correspondence should be called poor (overlap < 60% or insignificant) which is caused by the poor quality of the site maps.

1. The average overlap of ecotope group A12 with site type X12 is 43%. A fairly reasonable result after all, considering the difficulty of making a good map of site type X12 with only the help of the soil map, a map that does not distinguish between the various types of water (all waters are denominated 'water', including little isolated waters that are the domain of A12).
2. The average overlap of K23 with the site map amounts to 44%. This meagre result is due to the fact that the site type X23 can hardly be derived from the soil map, as the groundwater table class in the dune area is often absent on that map or indicated as 'very deep' (Gt VII), whereas, in reality, wet sites undoubtedly occur in the uneven terrain.

Table 4.8: Overlap of the ecotope maps with the corresponding site maps of Klijn *et al.* (1996). The column 'Expected' gives the overlap by pure chance (when there would be no relationship between the two types of maps). Statistical signifcance is calculated with a χ^2-test: nsig - not significant, $P < 0.01$; ** - very significant, $P < 0.001$ (after: Witte *et al.*, 1995).

	Completeness					Significance
	Low	High	Very high	Average	Expected	
K21	90	89	96	91	46	**
K22	93	92	99	93	54	**
K23	40	42	52	44	3	**
K27	96	98	99	97	81	**
K28	64	85	94	75	57	**
K41	100	100	100	100	48	**
K42	88	91	94	89	52	**
K43	57	84	91	74	11	**
K63	49	87	98	85	9	**
H22	93	92	100	94	54	**
H27	94	95	97	95	81	**
H28	73	62	85	74	57	**
H42	92	92	95	93	52	**
H43	60	84	97	81	11	**
H47	94	94	95	94	92	nsig
H63	66	83	98	88	9	**
A12	44	39	48	43	12	**
A17	99	99	99	99	80	**
A18	78	91	93	85	62	**

Besides, the soil map in areas outside the dikes, does not distinguish between areas that get flooded by seawater (K23 impossible) and those that do not (K23 possible).

3. Although the overlap H47 shows with the site map is high (94%), it cannot be called significant, since the X47-site map is not very specific: X47 is found in practically every single km-square.

4.6 Formal procedures for the determination of threshold values

4.6.1 Introduction

In this section I will explore to what extent the expert judgement, according to which the threshold values T of Table 4.7 have been determined, can be imitated by formal procedures. This may have several advantages: we may find out about the expert's motives as well as about the consistence of his statements. Equally, there is a chance that we will derive *general procedures* that can be of use for future threshold value determination. First, I will examine whether the threshold values listed in Table 4.7 can also be obtained by calculation (Section 4.6.2). Subsequently, I will examine whether there is a possibility of extrapolating the threshold values to grid-cells that are larger than km-squares (Section 4.6.3).

4.6.2 Calculation of threshold values for kilometer squares

To enable the prediction of T, let us now look for a variable offering a proper explanation for T. For this variable, the number of indicator species composing the ecotope group is of major importance: the more indicator species, the higher T. Furthermore, the way the group is composed of indicator values v (Table 4.4) has to be considered: the more a group is composed of species with high values of v, the fewer species are needed to detect it. If direct proportionality of T with the two factors mentioned (number of indicator species and average v) is taken for a fact, then T is also directly proportionate to the maximum indicator value score that is theoretically possible - $S_{\max,t}$, i.e. the sum of v of all indicator species from one ecotope group. However, the fact that maximum score in the gap-filled FLORBASE - $S_{\max,a}$ - may be significantly lower than $S_{\max,t}$, is neglected in this way. Fig. 4.16 shows, for various groups, the proportion between $S_{\max,a}$ and $S_{\max,t}$. This proportion, called *homogeneity* here, appears to vary considerably per group. Generally, groups that are extreme in nutrient-richness or acidity (very nutrient-rich/nutrient-poor, alkaline/acid) are fairly homogeneous, unlike the intermediate groups (moderately nutrient-rich, moderately acid).

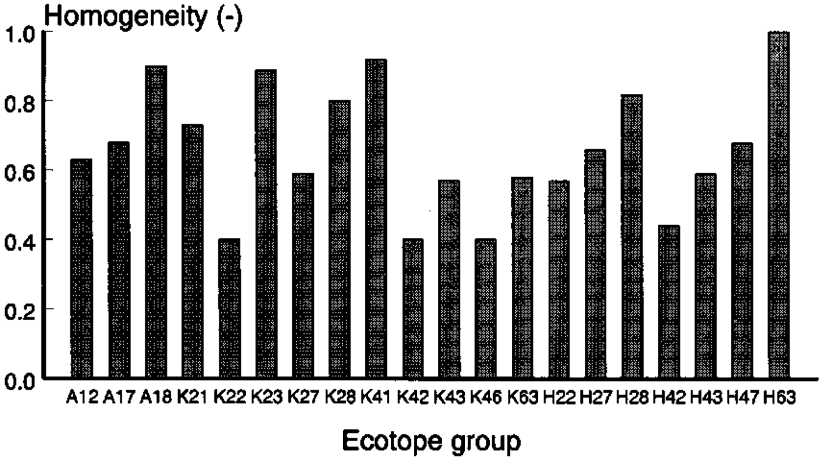


Figure 4.16: Homogeneity of 20 ecotope groups (after: Witte & Van der Meijden, 1993). Homogeneity is defined as the quotient of the maximum indicator value score found in FLORBASE-2c, and the theoretical maximum score, i.e. the sum of indicator values v.

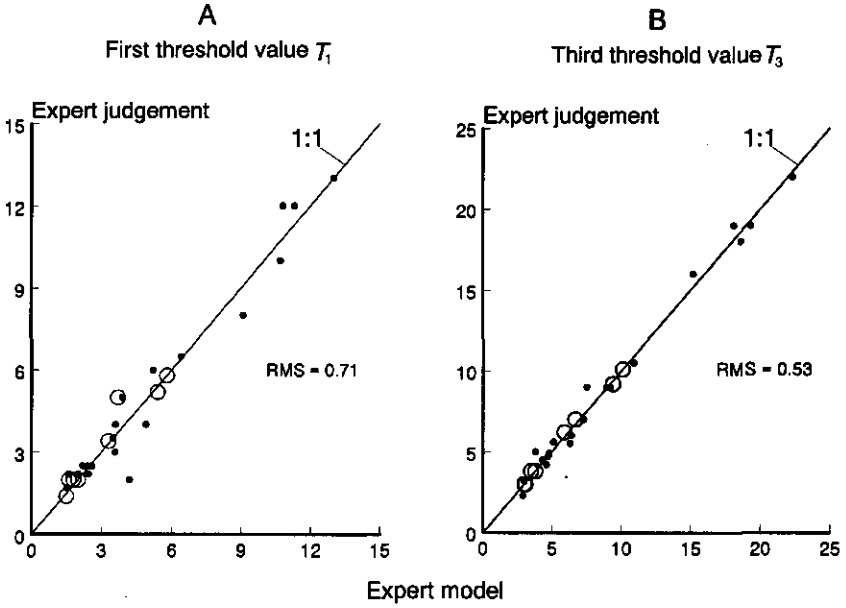


Figure 4.17: Threshold values of the expert versus the computed threshold values. The open circles are partly obtained with the aid of the expert model (Applying Eqs. (4.5) and (4.6) respectively), the closed ones are not.

If we also wish to consider this homogeneity, it is probably better to take the actual $S_{\max}$ as an explanatory variable. However, this measure is very chance-dependent: the occurrence of one outlier in FLORBASE already clouds the relation with T. Therefore, it is to be preferred to choose yet another high score, e.g. the one that is surpassed in no more than 0.2% of the number of km-squares. Assuming this 0.2%-score, we find the following linear relations:

$$\widehat{T}_1 = -0.03 + 0.43 S_{0.2} \qquad (r = 0.97) \tag{4.5}$$

$$\widehat{T}_3 = 0.49 + 0.72 S_{0.2} \qquad (r = 1.00) \tag{4.6}$$

where:

$\widehat{T}_1$ = estimated first threshold value
$S_{0.2}$ = 0.2%-score, dederived from km-squares of at least 50 species (Fig. 4.14)
$\widehat{T}_3$ = estimated third threshold value

These equations - to be called the *expert model* from now on - have been derived by Witte & Van der Meijden (1993) from the 20 ecotope groups, shown in Fig. 4.16. In Fig. 4.17 the T-values determined by expert judgement have been plotted against the values calculated by Eqs. (4.5) and (4.6). Equally, T-values partly estimated on the basis of these equations, are shown by open circles. These T-values refer to the 8 groups not shown in Fig. 4.16.

Fig. 4.17 shows that *the judgement was carried out very consistently.* With a view to the application of distribution maps in policy studies this is a gratifying result. As ecotope groups are unevenly distributed, an inconsistent judgement would imply the risk of certain regions dominating the results of these studies, whereas other regions would not get the valuation they deserve. Fig. 4.17 also shows that *the judgement can be imitated* by Eqs. (4.5) and (4.6) and that all the expert needs to do - if anything - is making small corrections in the outcomes.

For the lower values of T_1, relatively large deviations from the model may be observed (Fig. 4.17A). For some outliers, T_1 has been modified on the basis of Eq. (4.5), but this did not improve the ecotope map concerned. Apparently, also other factors should be considered when determining T_1.

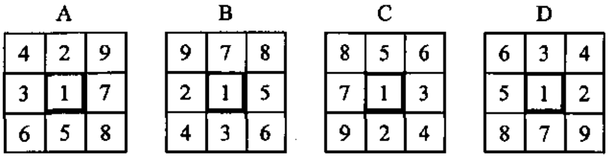
4.6.3 Extrapolation of threshold values to other grid sizes

The T-values from Table 4.7 can be used for grid-cells measuring 1 × 1 km, but not for cells of a different size (e.g. hour squares). This section serves to examine how the T-values may be extrapolated to larger cells. This question is for instance important for the analysis of distribution records of wild plants that were found in the inventory period 1902 – 1950. As was described in Section 1.4.2, the grid-cells in that inventory period measure 1.25×1.04 km (1.30 km^2). They are sometimes called *quarter squares*, since they measure a quarter of a pre-1950 hour square.

The larger the grid-cells, the larger the number of found species - and their derived indicator value scores S - may be expected to be. Consequently, larger cells require larger T-values. Moreover, the relation between S and the grid-cell size depends on the height of the score. For it may be assumed that for cells with a very high value of S, the score hardly increases when the cells are enlarged. For cells yielding a particularly low value of S, on the other hand, S is bound to increase substantially as the grid-cells are enlarged. In other words, the frequency distribution of S over the cells will become more even as the cells are enlarged, mainly because of the fact that cells of low S-values disappear.

The influence the grid-cell size has on S has been examined on the basis of data from the gap-filled FLORBASE. For every km-square it has been determined how S increases when neighboring km-squares are added. For this, the calculation schemes of Fig. 4.18 have been applied, starting from scheme A.

First the *starting score* S_1 in centre square 1 is determined, then square 2 is merged with this centre square, and so on, until all 9 squares are merged. The following three squares are handled in a similar way, but this time with the schemes B, C and D respectively. Subsequently, for the following km-quare, scheme A is taken up and the procedure starts all over again. This is repeated again and again, until the whole of the flora database is run through. The analysis is carried out with alternating calculation schemes in order to prevent artificial results in



A

4	2	9
3	1	7
6	5	8

B

9	7	8
2	1	5
4	3	6

C

8	5	6
7	1	3
9	2	4

D

6	3	4
5	1	2
8	7	9

Figure 4.18: Scheme for the assessment of the relation between grid-size and indicator value score S. Each cell represents 1 km-square.

the case of ecotope groups having a linear distribution pattern (e.g. K63 in the dunes and K46 along the rivers).

Finally, average scores are calculated from the km-results for each of the nine combinations of squares. Fig. 4.19 shows - for various starting scores S_1 - the results, as calculated for ecotope group K27. Especially when S_1 is low, a relatively large increase is reached. The classes of S_1 in Fig. 4.19 have been chosen in such a way that at least all T-values for FLORBASE are comprised. For K27 the three T-values amount to 12.0, 15.5 and 19.0 (Table 4.7). They are accompanied by the S_1-classes A, C and E (Fig. 4.19).

The curves of Fig. 4.19 are derived by linear regression. They are of the following exponential equation through the point $(1, 1)$:

$$\frac{\widehat{S_x}}{S_1} = x^c \tag{4.7}$$

where:

$\widehat{S_x}$ = estimated score in x km-squares
S_1 = starting score
x = number of aggregated km-squares
c = exponent

Values for exponent c are mentioned with the curves in Fig. 4.19. The curves may for instance be used for the calculation of T-values for quarter squares. For this, the values of Table 4.7 - meant for km-squares -are multiplied with 1.30^c. The first threshold value of K27 (12), for instance, falls within class A ($c = 0.257$), which means that it has to be multiplied by $1.30^{0.257} = 1.07$ (the result is a T-value of 12.8)[3].

However, there is the possibility of a more elegant interpolation technique, which has the advantage of resulting in a generally applicable equation between S and x. This technique will be illustrated by K27 as well.

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[3] An interesting consequence of the relatively large increase of S when grid-cells have low starting scores S_1, is that it implies the existence of a grid-cell, of which the size is such that the extrapolated first and third threshold values are equal. Above this grid-size, it is no longer possible to distinguish between completeness classes!

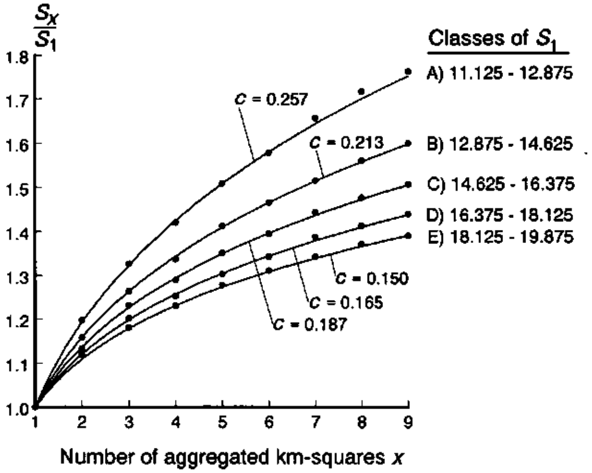


Figure 4.19: Score S_x of x km^2 - relative to the starting score S_1 of 1 km^2 - versus the number of aggregated km-squares x (ecotope group K27). With the aid of the classes A-E, distinction is made in S_1. The drawn curves are formed by $\widehat{S_x}/S_1 = x^c$.

In Fig. 4.20A, the exponents c for ecotope group K27 have been plotted against the average scores of the S_1-classes of Fig. 4.19. The following exponential function was fitted through the points by linear regression:

$$\widehat{c} = aS_1^b \tag{4.8}$$

Where a and b are fitting parameters.

Substitution of Eq. 4.8 in Eq. 4.7 results in the expression of S as a function x:

$$\widehat{S}_x = S_1 x^{aS_1^b} \tag{4.9}$$

Eq. 4.9, as well as experimental values of a and b, now permit us to calculate the relation between S_x and x for every ecotope group. In Fig. 4.20B this has been done for K27 with $x = 1.30$, so that the relation between S_1 and $S_{1.30}$ is shown. Here, Eq. 4.9 appears to result in a practically straight line.

In the previous section (4.6.2), it was argued that T-values for FLORBASE have been determined in a strikingly consistent way. Now that we know the relation between x and S_x it is very tempting to test the consistency of the experts' judgement *in space and time.* Nothing prevents us from doing so, for in 1989 T-values have been determined by Van der Meijden for 12 ecotope groups for the ATLAS database's second observation period (1950 – 1980) with hour squares of 25 km^2 (Witte & Van der Meijden, 1989; Witte & Van der Meijden, 1990). With Eq. 4.9 we calculate the threshold values on hour square level, upon which we compare the results with Van der Meijden's judgement in 1989. However, two

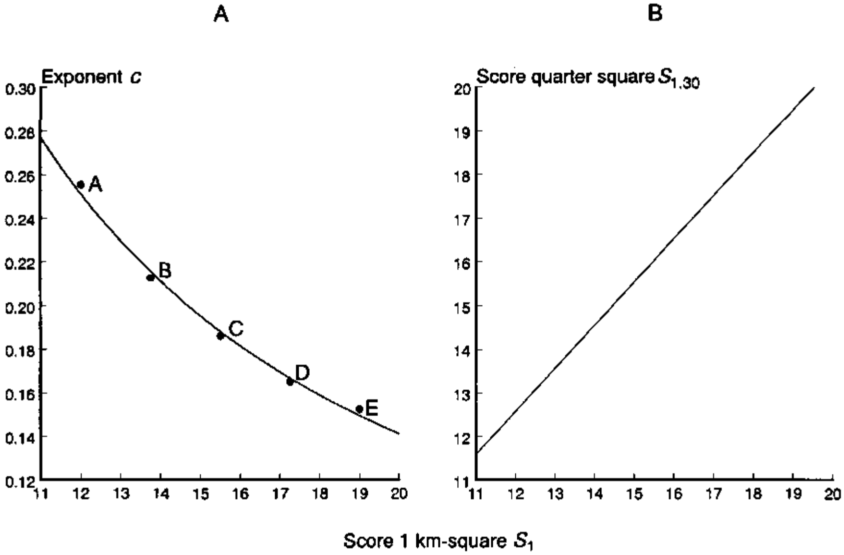


Figure 4.20: Relation between the score of K27 in 1 km^2 - S_1 - and: A - exponent c of Eq. (4.7); B - the K27-score per quarter square $S_{1.30}$.

methodical differences between the determination of T-values for the ATLAS database and that of FLORBASE hamper the comparison:

- The score calculation for the ATLAS database has been carried out in a different way from that for FLORBASE. For the ATLAS database a different list of indicator species was used; besides, every species was given an equal indicator value $v = 1$, whereas the FLORBASE method assigned v-values varying from 0.33 to 1.00.
- The T-values for FLORBASE refer to a gap-filled database, as opposed to the threshold values for the ATLAS database.

These differences may be compensated as follows:

- First new indicator value scores are calculated for the ATLAS database, i.e. scores that are based on the same indicator species and v-values as were used for FLORBASE. Subsequently, it is calculated what the T-values should amount to in order to have the same number of hour squares fall within a certain completeness class (noise, low, high and very high) as was the case on the old distribution maps. In this way, the old T-values have been transformed into new T-values.
- As gap-filling mainly affects the low S-values, we only incorporate T_3 in the comparison.

In Fig. 4.21B T_3, extrapolated to hour squares, has been plotted against the translated experts' judgement from 1989. To give an insight in the degree of extrapolation, the original FLORBASE T_3-values are shown in Fig. 4.21A. In spite of the different methods and the conversions, the extrapolated values appear to

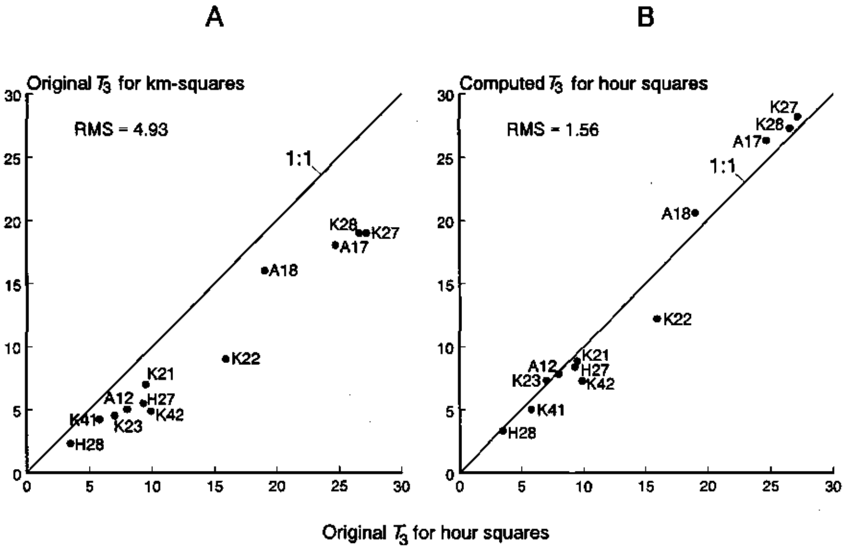


Figure 4.21: Third threshold values T_3 for km-squares (A) and computed T_3-values for hour squares (B) versus T_3-values for hour squares obtained by expert-judgement.

correspond very well with the experts' judgement of 1989. It may be concluded that the judgement on km-square level is convertible to hour squares for the observation period 1950 – 1980. Not only does this result once more *affirm the consistency in the experts'* judgement, it also *emphasizes the soundness of the proposed extrapolation method.*

4.7 Discussion

The DEMNAT model works with spatial entities ('ecoplots', Section 2.3.4), which are partly based on the ecotope maps. As a consequence, it is important that indicator species, instead of being randomly scattered within a km-square, occur together within a reasonable distance: in an actual vegetation. For the determination of the threshold values this hypothesis was a starting-point. I conclude that this hypothesis is now affirmed by the comparison of the threshold value method with vegetation relevés (Section 4.5.2) and with other maps about the natural environment in the Netherlands (Section 4.5.3). However, we do not yet have information about the reliability of the threshold value method. In practice, the comparison with other maps is difficult, as we have to cope with different mapping purposes, with different chart indices, and with all sorts of chart inaccuracies.

In order to find out about the reliability of the threshold value method, I propose to select km-squares of which the flora has been thoroughly examined and of which many vegetation relevés are available as well. These relevés have to be assigned to ecotope groups with a method specially developed for this research (e.g. a method that is developed on the basis of ECOTYP, Section 4.5.2) and have to be given a botanical quality indication. The results may then be compared to those of the threshold value method, applied to the inventory data of entire km-squares.

Although it looks as if the threshold value method can detect actual vegetations, theoretically the indicator species of a certain ecotope group may also be distributed all over the km-square. In that case, the completeness, as indicated by the maps, would indicate a large area of the ecotope group (*quantity*), instead of a relatively species-rich vegetation (*quality*). It may be doubted whether this makes any difference for the valuation of km-squares. But Witte & Van der Meijden (1993) make it clear that quality and quantity are correlated: a high completeness indicates a relatively large ecotope group. If we take hour squares for instance, understanding the km-squares within those hour squares as locations, then it may be shown that a high indicator value score points at a species-rich location (high completeness in at least one of the 25 km-squares), as well as a large area of the ecotope group (several km-squares within the hour square where the group concerned is found). The fact that completeness gives an indication about the size of the ecotope group is illustrated by the geographical clustering on the ecotope maps of completeness classes: the completeness in a certain km-square gives an indication about the completeness in a neighboring km-square. For species such clustering was shown in several studies (Brown, 1984) and it may easily be visualized: a cluster of hour squares of a particular species often means that, within those hour squares, several km-squares containing that species are found, whereas isolated hour squares often contain only one or several km-squares containing the species (see e.g. Fig. 1.5). As for ecotope groups, the clustering can be seen in their distribution maps: km-squares, qualified as 'very high' are often situated in the vicinity of one another.

I estimate that the size of species-rich vegetations varies from minimally some hundreds of square meters to - depending on the ecotope group - maximally several hectares (A12) to dozens of hectares (K28). However, this estimation is based on my personal field experience and not on hard facts. Klijn (1988) mentions for ecotope types an indicative mapping scale of $1:5,000 - 1:2,5000$, with a corresponding basic mapping unit of $0.25 - 1.5$ ha.

An objection against the threshold value method could be that intermediary vegetations, containing indicator species from more than one ecotope group, would not be fully represented on the distribution map. A well developed vegetation, for instance, with characteristics of K27, as well as of K28, would on both maps be classified as 'moderate'. For assessment of conservation values of km-squares, this is not necessarily a problem, but the objection becomes serious if a moderately developed intermediary vegetation would appear on neither of the two maps. But, I expect that this objection will hardly turn out to be a real problem, as the ecosystem classes have been determined so broadly that most vegetations are bound to fall within their borderlines. However, if anyone would wish to investigate the gravity of the objection, the earlier mentioned reliability-test of the threshold value method could be of use.

Another comment to be given on the maps is that the indicated completeness does not necessarily signify *homogenous* vegetations, i.e. vegetations exclusively containing species from one and the same ecotope group. In the case of some ecotope groups, the completeness could also signify the existence of vegetations with a broad spectrum of ecological groups. Such is for instance often the case with vegetations that are characterized as K22. Well developed quagfens, for instance, contain species from at least 5 ecotope groups - A12, A17, K21, K22 and K27 - which means that they have a *characteristic heterogeneity*. Nevertheless it is preferable to characterize them as 'wet, nutrient-poor and weakly acid', firstly because the number (not necessarily the cover!) of K22-species dominate in such a quagfen, and secondly because it is in particular the K22-species that make the quagfen vegetation differ from other vegetations. The distribution maps of several other ecotope groups are likely to show vegetations containing species from the same ecological group. This is for instance the case with ecotope group K21, which mostly comprises wet heathlands and bogs.

In Section 4.6.2, it was shown that the judgement serving the determination of the threshold values has been very consistent and lends itself to imitation by a calculation model. Although this result sustains the judgement's reliability, it remains disputable to entrust a normative issue, such as the classification of km-squares in terms of 'low', 'high' and 'very high', to one single person. Therefore, the judgement of other experts should also be examined. I propose the following procedures.

Firstly, one possibility is the application of the expert model of T-values (Eqs. (4.5) and (4.6)) to other groups of species. The resulting distribution maps should be acceptable for other experts. This method was applied by Runhaar & Van 't Zelfde (1996). They used the expert model to calculate T-values for the ecosystem types of Bal *et al.* (1995). The distribution maps they then derived from those types with the help of FLORBASE, were qualified by them as genuine (J. Runhaar, personal communication). In this way, the expert judgement could also be applied on foreign flora databases of other groups of species. To this end, Eqs. (4.5) and (4.6) would first have to be adapted to the larger foreign grid-cells (Section 4.6.3 gives instructions for a way to do this).

A second and more straight way is to have experts determine T-values themselves, upon which this judgement is compared to the expert model. Currently, experts from the Institute of Nature and Forestry Research (IBN-DLO) are determining T-values for the plant sociological units of Schaminée *et al.* (1995). I hope to be able to make 'models' from these experts, and to compare them with each other, as well as with the expert model from Section 4.6.2 (Eqs. (4.5) and (4.6)).

The third possibility I would like to mention is a somewhat indirect and rough method. The 28 ecotope maps are combined into one map, showing the botanical conservation value of km-squares. Subsequently, experts are requested to state whether or not they can agree to that map. This method will be discussed in the following chapter (Chapter 5).

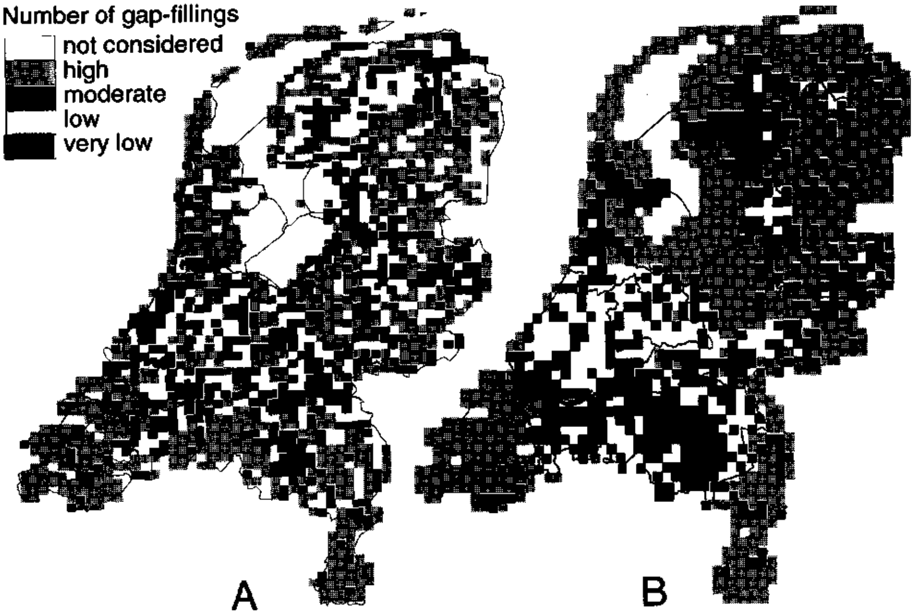


Figure 4.22: Number of gap-fillings of the ATLAS database: A - inventory period 1902 – 1950; B - inventory period 1950 – 1980. The number of gap-fillings gives an indication of the intensity of the inventory: the less gap-fillings, the higher the inventory intensity. After: Witte & Van der Meijden (1993).

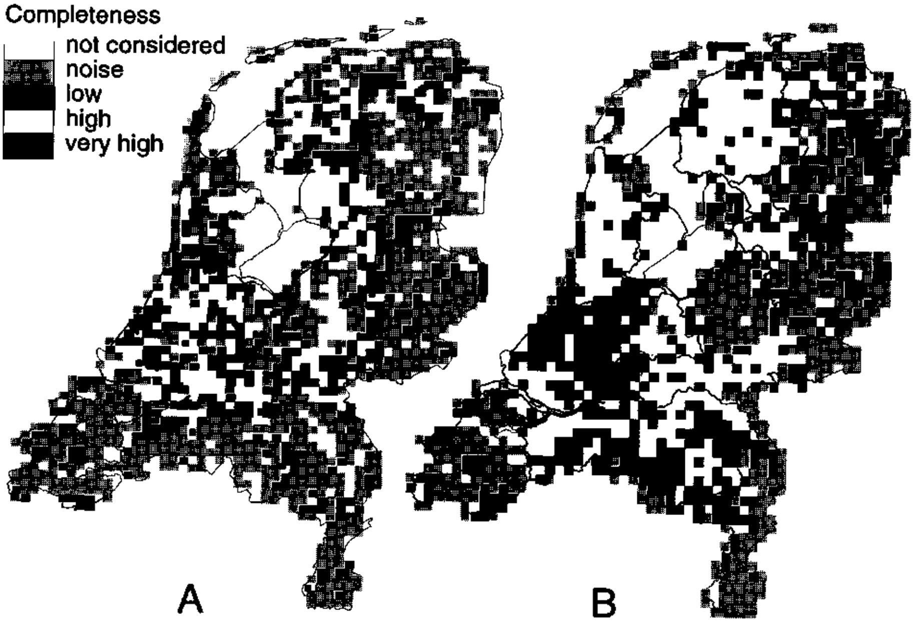


Figure 4.23: Distribution of ecotope group A18 (vegetation in stagnant, very nutrient-rich water; ecotope types V18 and W18) on the basis of the ATLAS data base: A - 1902 – 1950; B - 1950 – 1980. The absence of adequate footwear is likely to be the explanation for much of the 'progress' shown on this map: the progress is probably largely an artefact. After: Witte & Van der Meijden (1993).

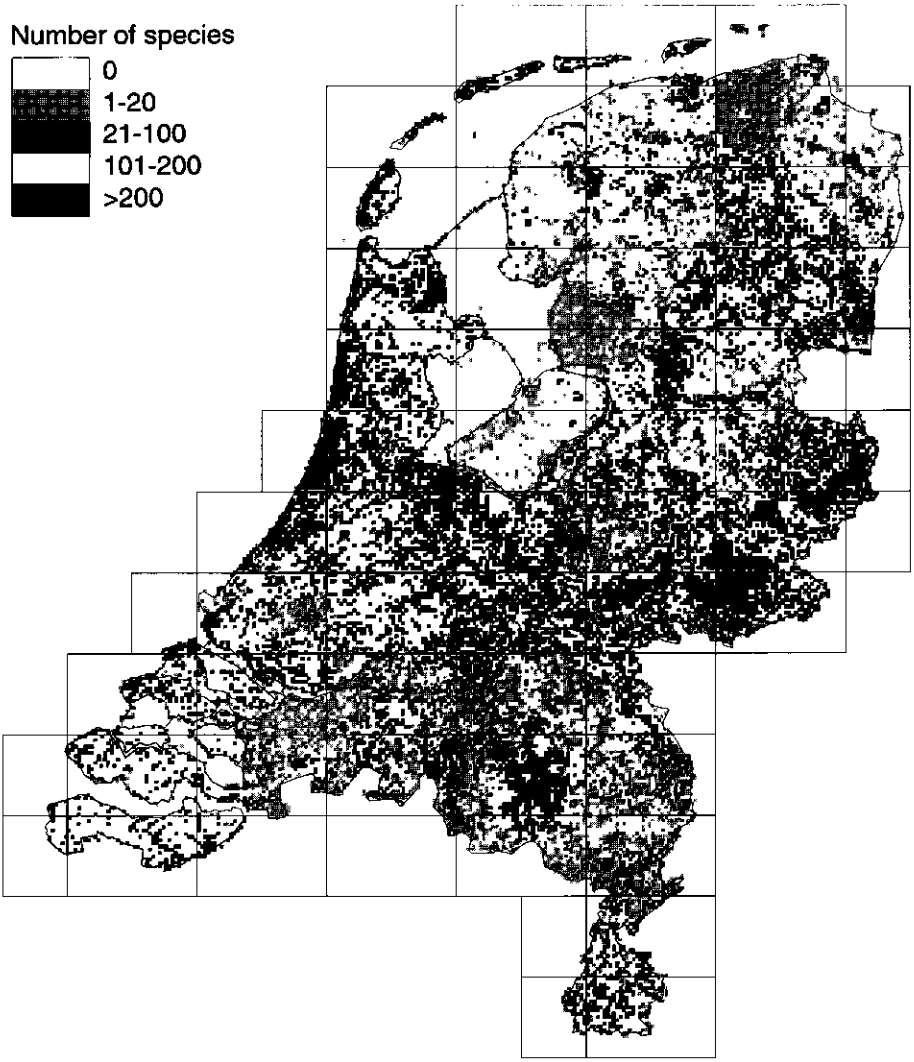


Figure 4.24: Number of species per km-square in FLORBASE-2c (without gap-filling).

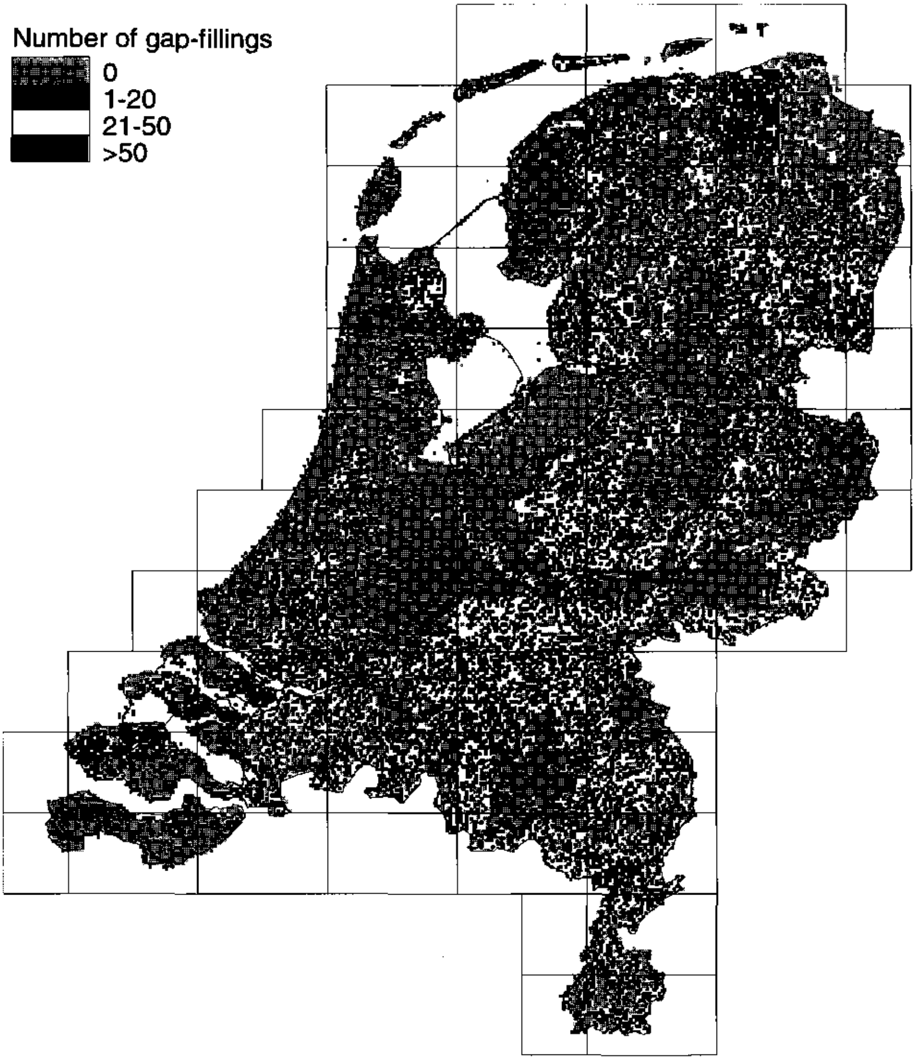


Figure 4.25: Increase in the number of species per km-square of FLORBASE-2c as a result of gap-filling.

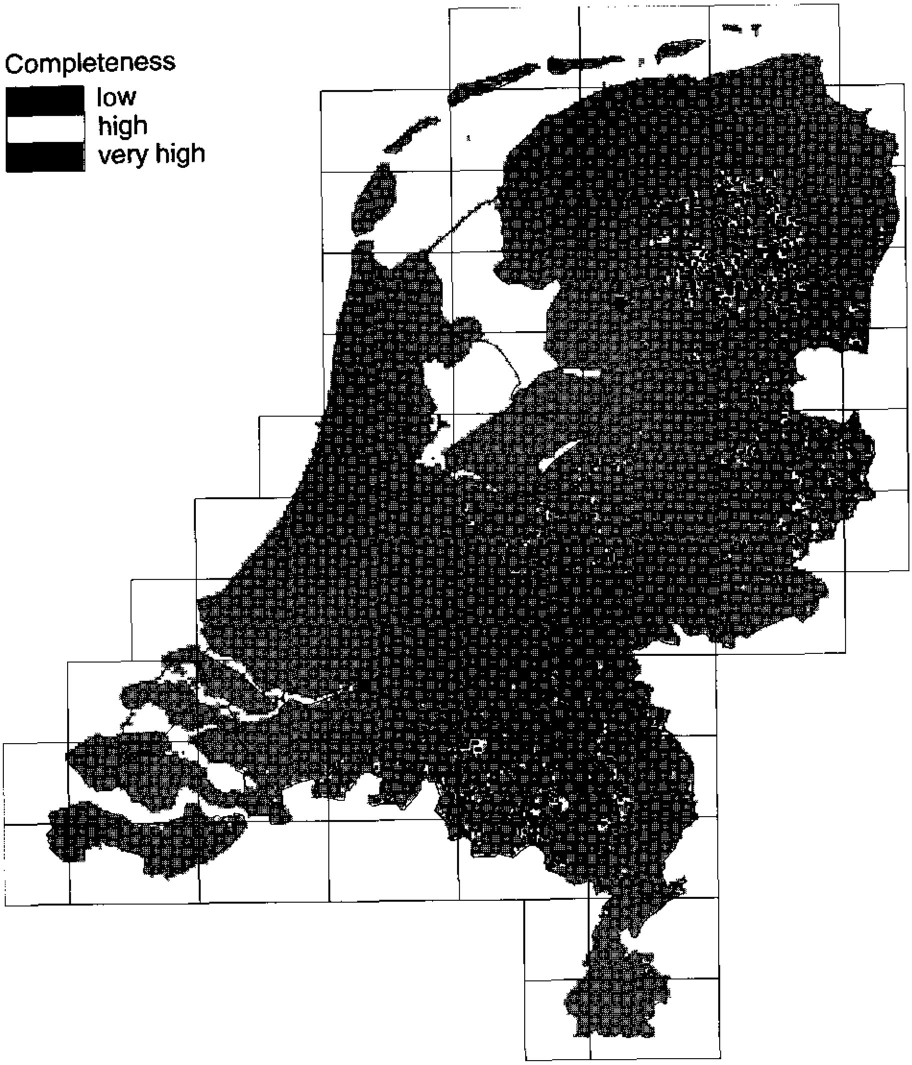


Figure 4.26: Distribution of ecotope group K21: pioneer vegetations and grasslands on a wet, nutrient-poor, acid soil (ecotope types P21, G21). By nature, there are less indicator species on the isles of the Wadden sea and in the 'Laagveendistrict', than there are in the Pleistocene flora districts. This probably explains the low completeness in these areas. It is remarkable that this ecotope group is poorly represented on the 'Utrechtse Heuvelrug'. Based on FLORBASE-2c (gap-filled).

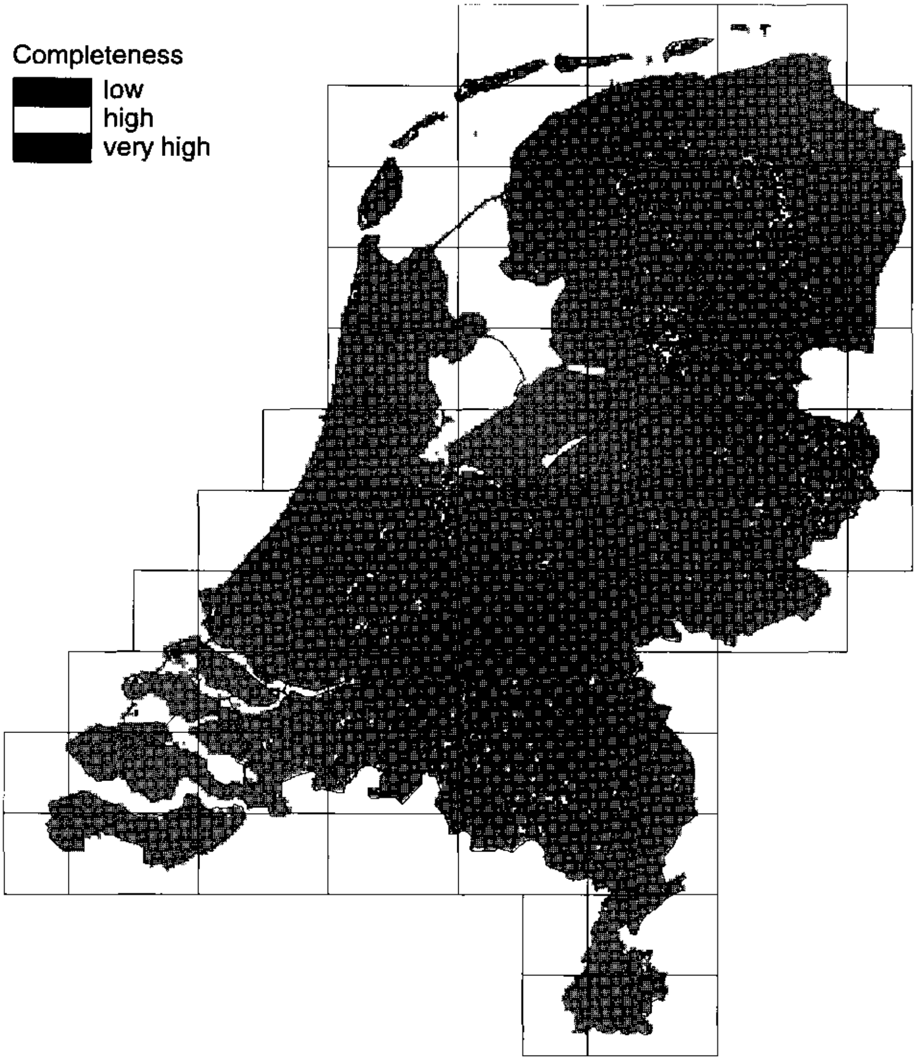


Figure 4.27: Distribution of ecotope group K22: pioneer vegetations and grasslands on a wet, nutrient-poor, neutral soil (ecotope types P22, G22). This ecotope group is almost completely restricted to nature reserves. Based on FLORBASE-2c (gap-filled).

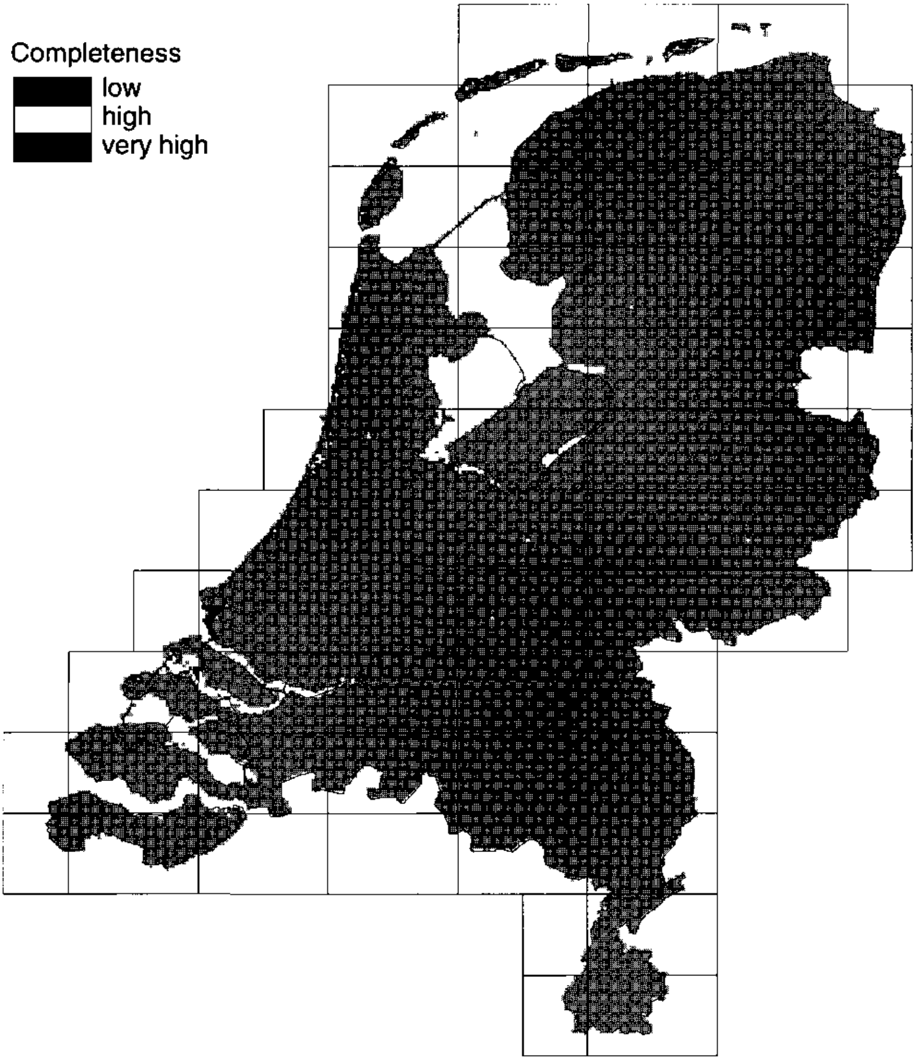


Figure 4.28: Distribution of ecotope group K23: pioneer vegetations and grasslands on a wet, nutrient-poor, alkaline soil (ecotope types P23, G23). This ecotope group is entirely restricted to nature reserves. Based on FLORBASE-2c (gap-filled).

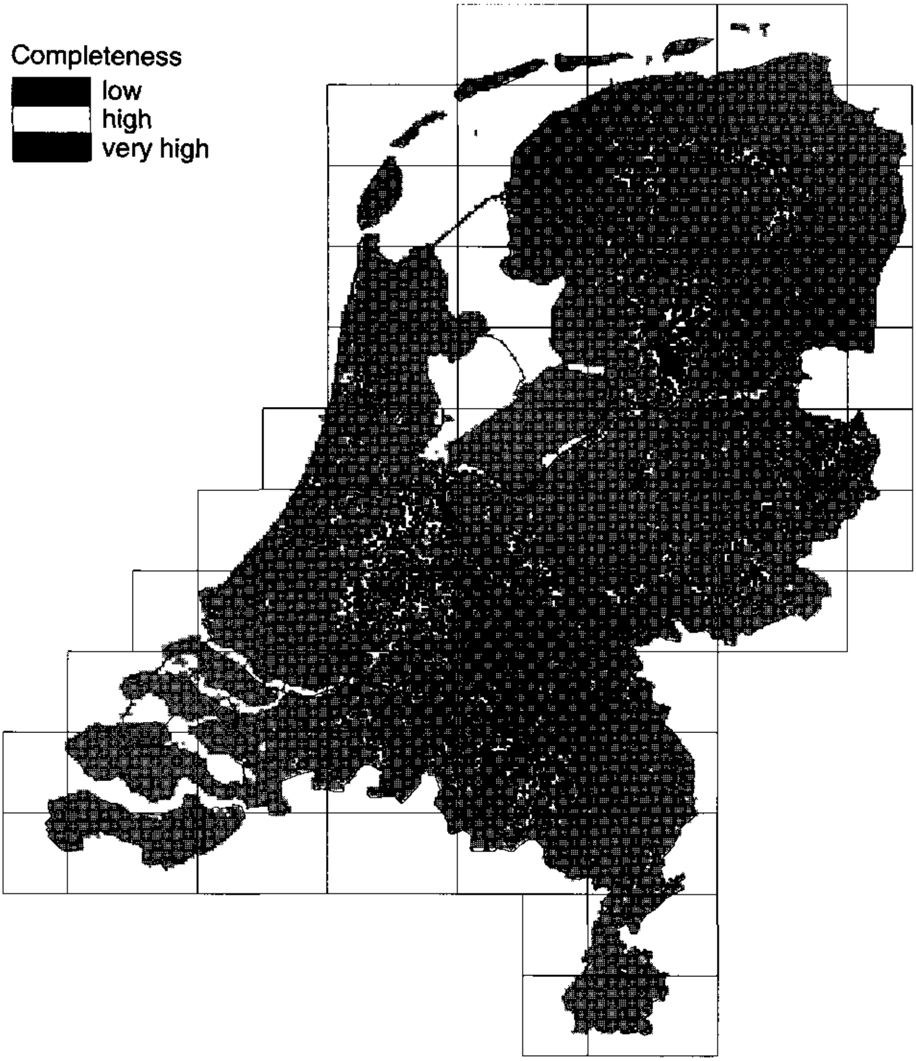


Figure 4.29: Distribution of ecotope group K27: pioneer vegetations, grasslands and tall herbaceous vegetations on a wet, moderately nutrient-rich soil (ecotope types P27, G27, R27). This ecotope group is largely bound to brook valleys and the 'Laagveendistrict'. Based on FLORBASE-2c (gap-filled).

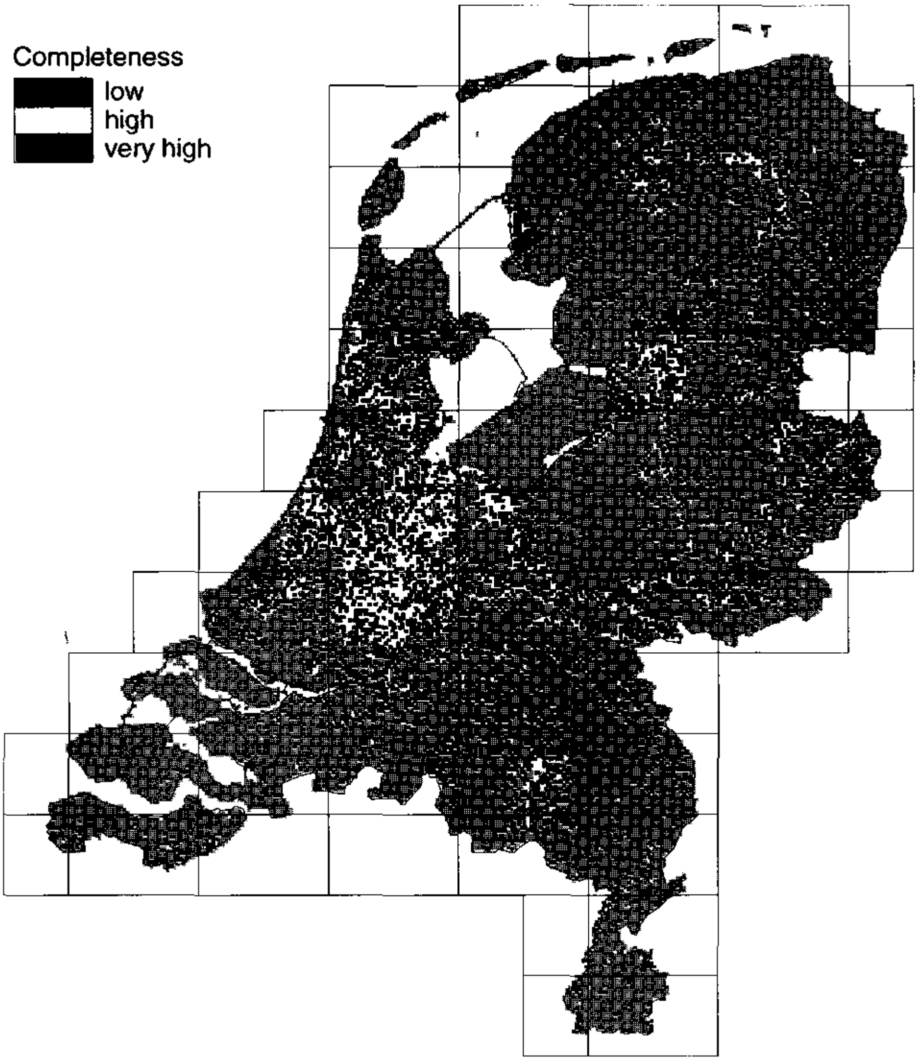


Figure 4.30: Distribution of ecotope group K28: pioneer vegetations, grasslands and tall herbaceous vegetations on a wet, very nutrient-rich soil (ecotope types P28, G28, R28). This ecotope group is especially found in the 'Laagveendistrict' and the river area. Its occurrence outside these areas may be an indication of the inlet of river water or of the influence of excessive manuring. Based on FLORBASE-2c (gap-filled).

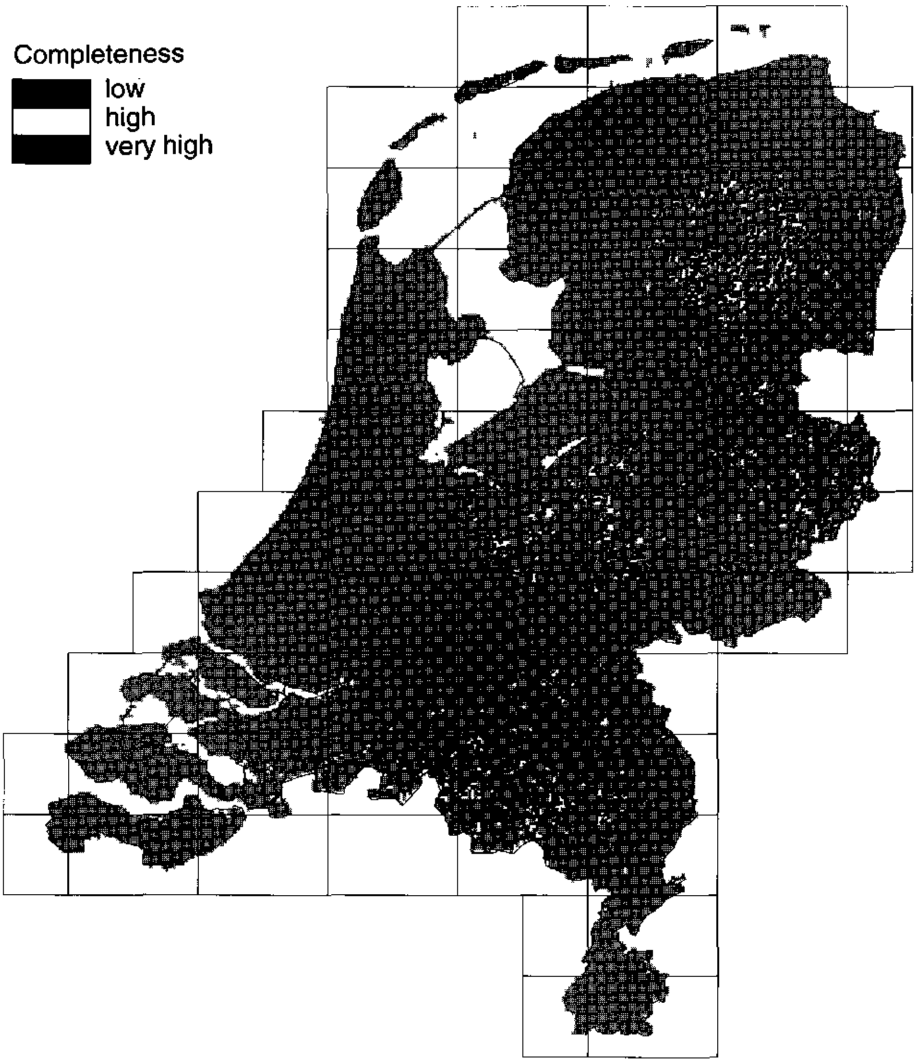


Figure 4.31: Distribution of ecotope group K41: pioneer vegetations and grasslands on a moist, nutrient-poor, acid soil (ecotope types P41, G41). As the threshold values T of this ecotope group are close to each other, the reliability of this map is low. By nature, there are less indicator species on the isles of the Wadden sea and in the 'Laagveendistrict', than there are in the Pleistocene flora districts. It may be that the completeness on the map is too low in these areas. Based on FLORBASE-2c (gap-filled).

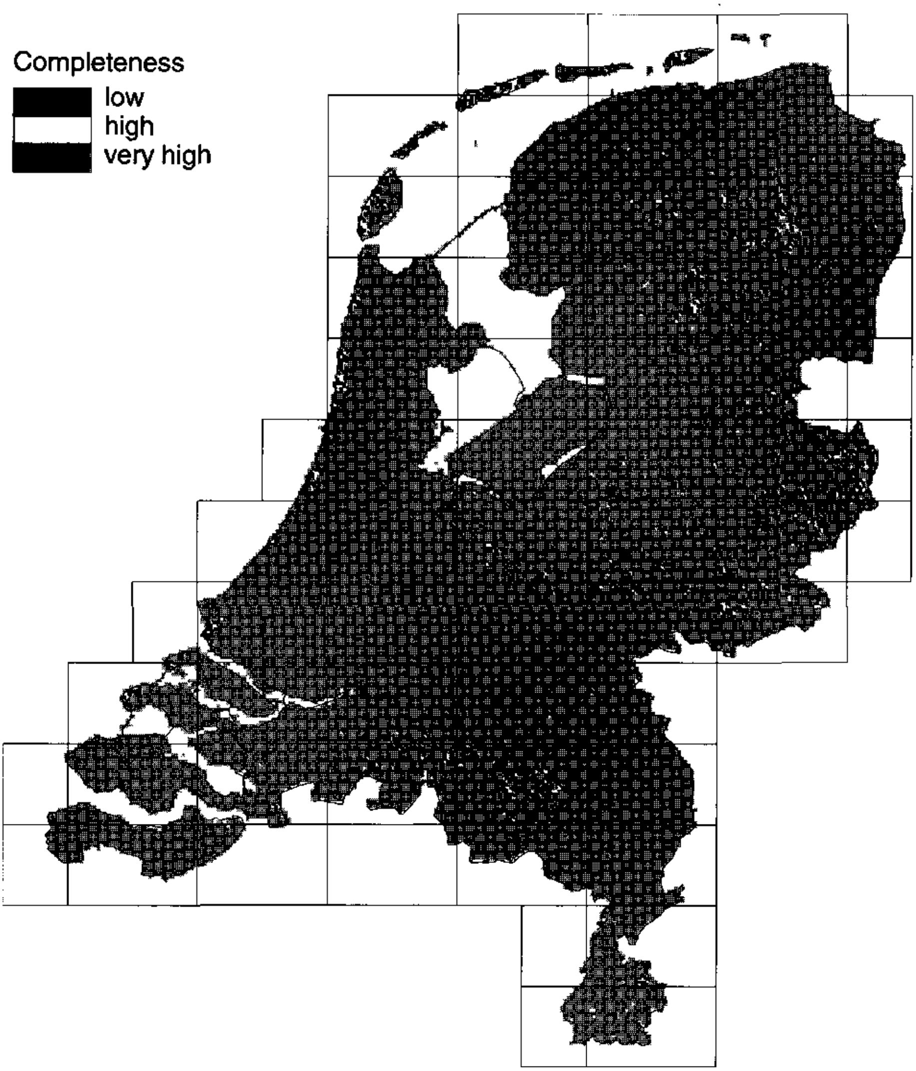


Figure 4.32: Distribution of ecotope group K42: pioneer vegetations and grasslands on a moist, nutrient-poor, neutral soil (ecotope types P42, G42). This ecotope group is almost completely restricted to nature reserves. Based on FLORBASE-2c (gap-filled).

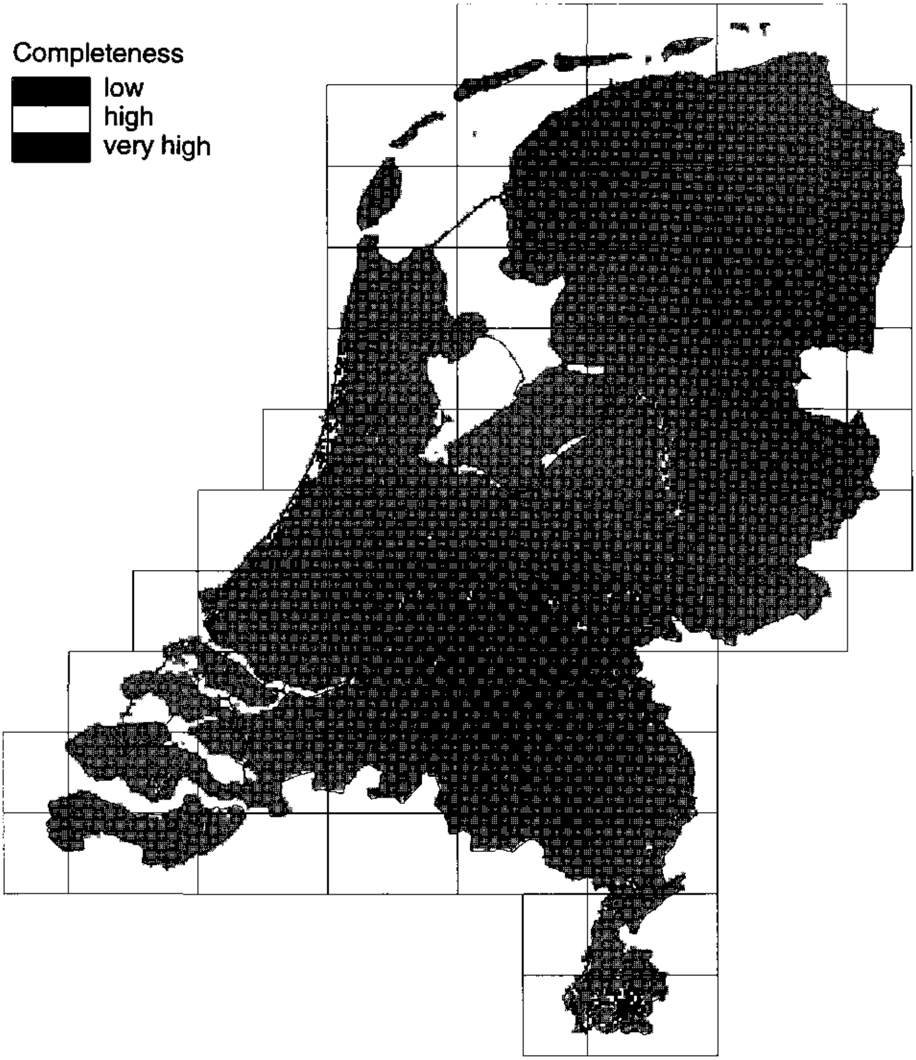


Figure 4.33: Distribution of ecotope group K43: pioneer vegetations and grasslands on a moist, nutrient-poor, alkaline soil (ecotope types P43, G43). This ecotope group is especially found in the southern part of the province of Limburg. Here it occurs in nature reserves and - fragmentary - in verges. Based on FLORBASE-2c (gap-filled).

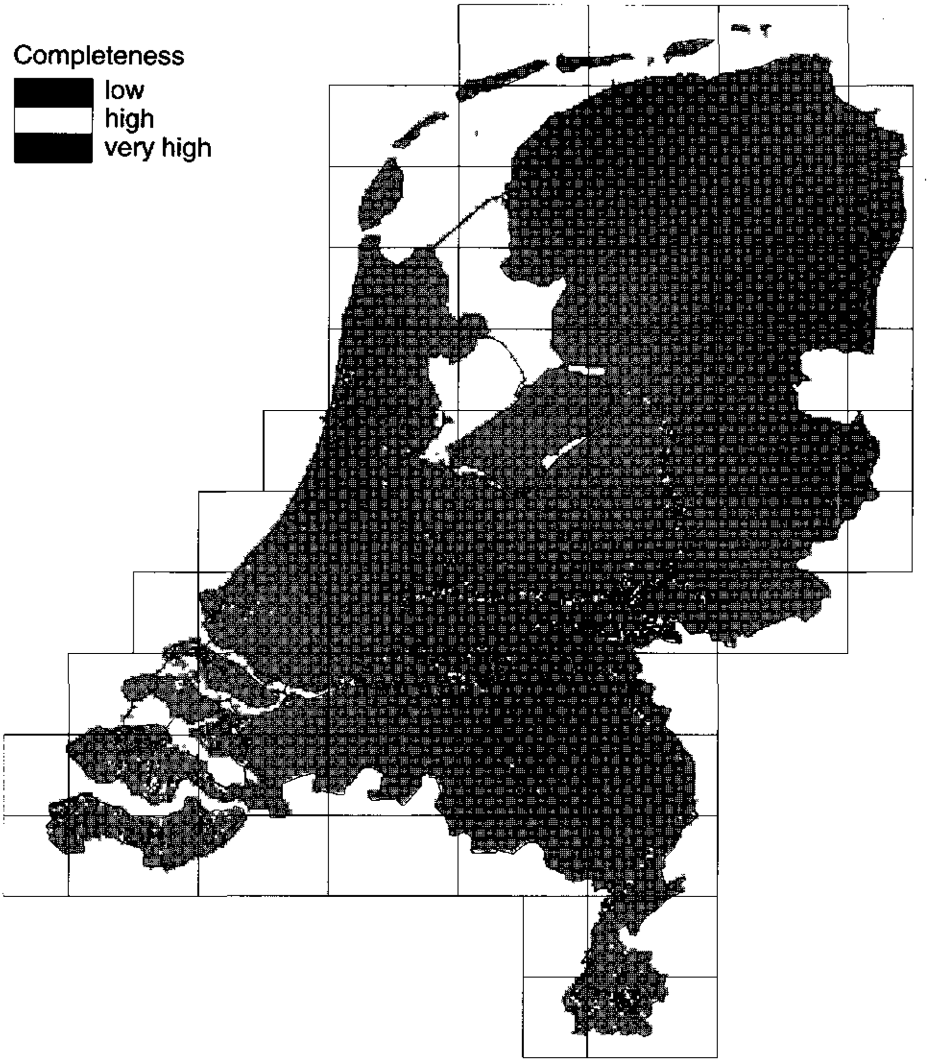


Figure 4.34: Distribution of ecotope group K46: pioneer vegetations, grasslands and tall herbaceous vegetations on a moist, moderately nutrient-rich, alkaline soil (ecotope types P47*, G47*, R47*). This ecotope group is especially found on river dikes, on dikes in the province of Zeeland and in the southern part of the province of Limburg. Based on FLORBASE-2c (gap-filled).

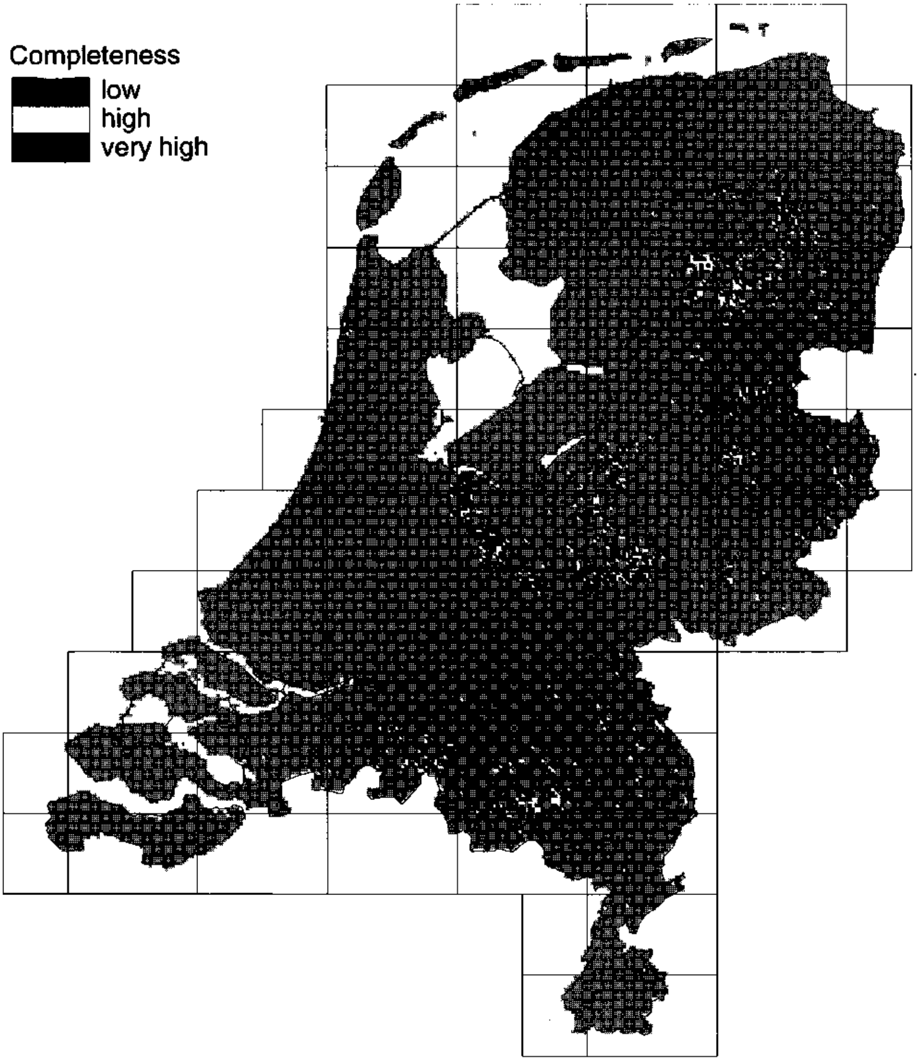


Figure 4.35: Distribution of ecotope group K61: pioneer vegetations and grasslands on a dry, nutrient-poor, acid soil (ecotope types P61, G61). As the threshold values T of this ecotope group are close to each other, the reliability of this map is low. By nature, there are less indicator species on the isles of the Wadden sea, than there are in the Pleistocene flora districts. It may be that the completeness on the map is too low on these isles. Based on FLORBASE-2c (gap-filled).

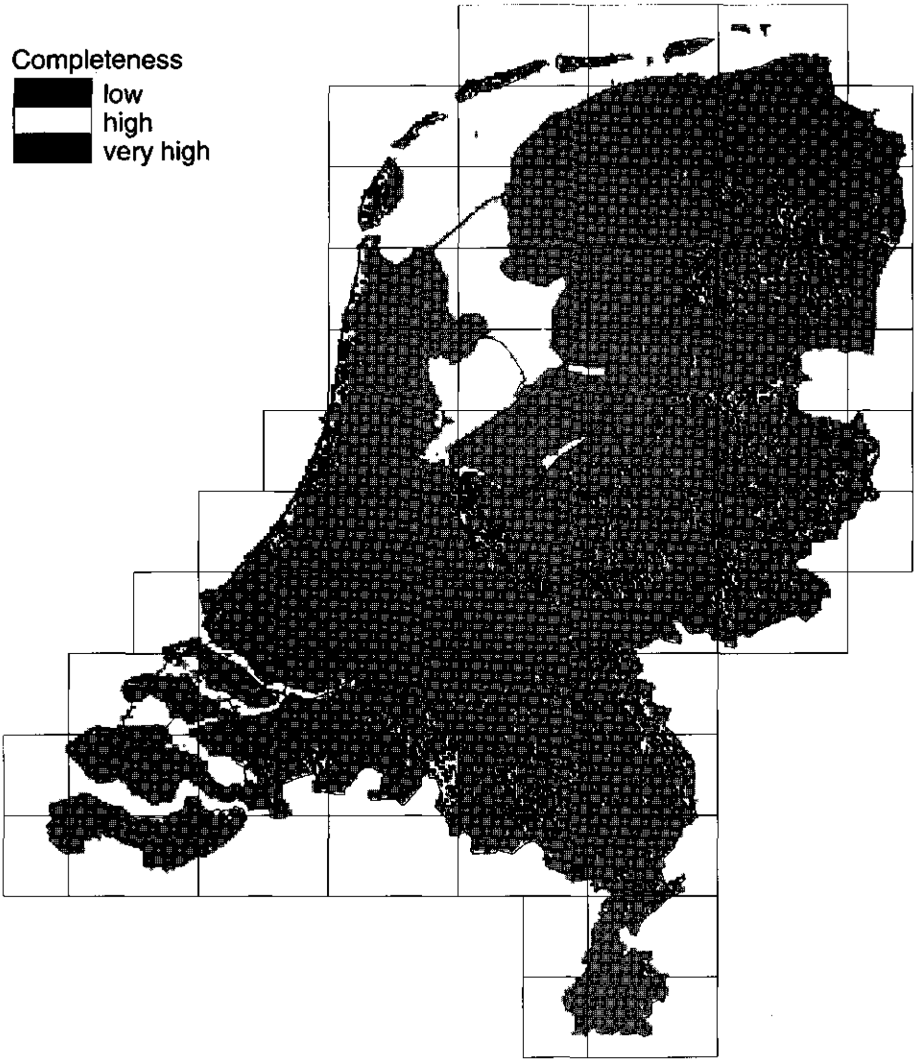


Figure 4.36: Distribution of ecotope group K62: pioneer vegetations and grasslands on a dry, nutrient-poor, neutral soil (ecotope types P62, G62). Based on FLORBASE-2c (gap-filled).

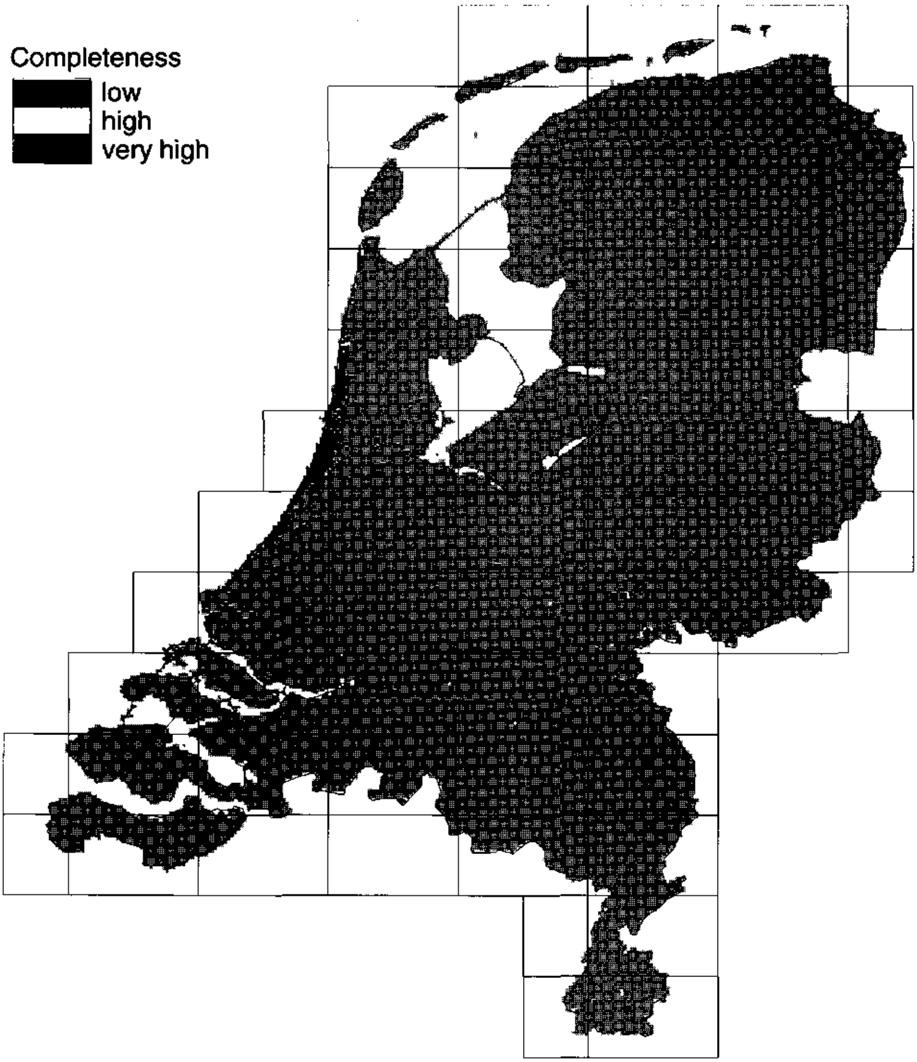


Figure 4.37: Distribution of ecotope group K63: pioneer vegetations and grasslands on a dry, nutrient-poor, alkaline soil (ecotope types P63, G63). This ecotope group is almost completely restricted to nature reserves in the calcareous dune area. Based on FLORBASE-2c (gap-filled).

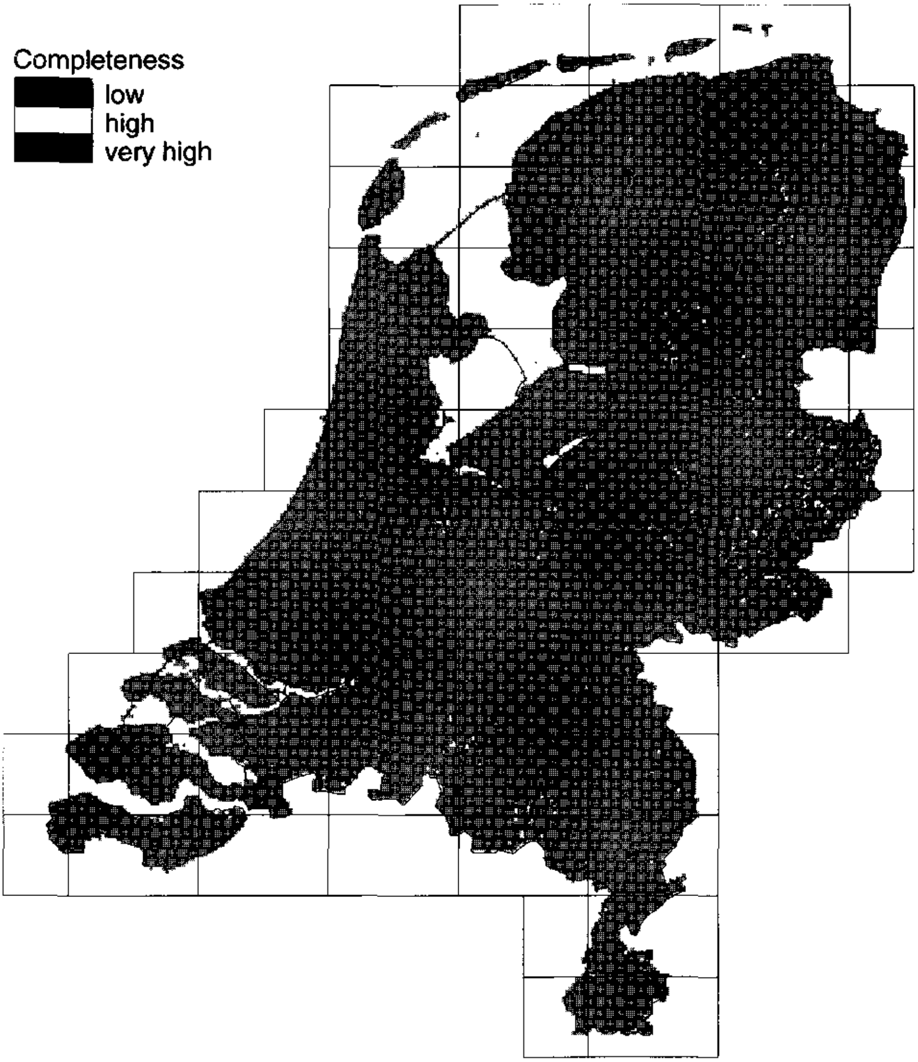


Figure 4.38: Distribution of ecotope group H22: woods and shrubs on a wet, nutrient-poor, neutral soil (ecotope type H22). This ecotope group is almost completely restricted to nature reserves. Based on FLORBASE-2c (gap-filled).

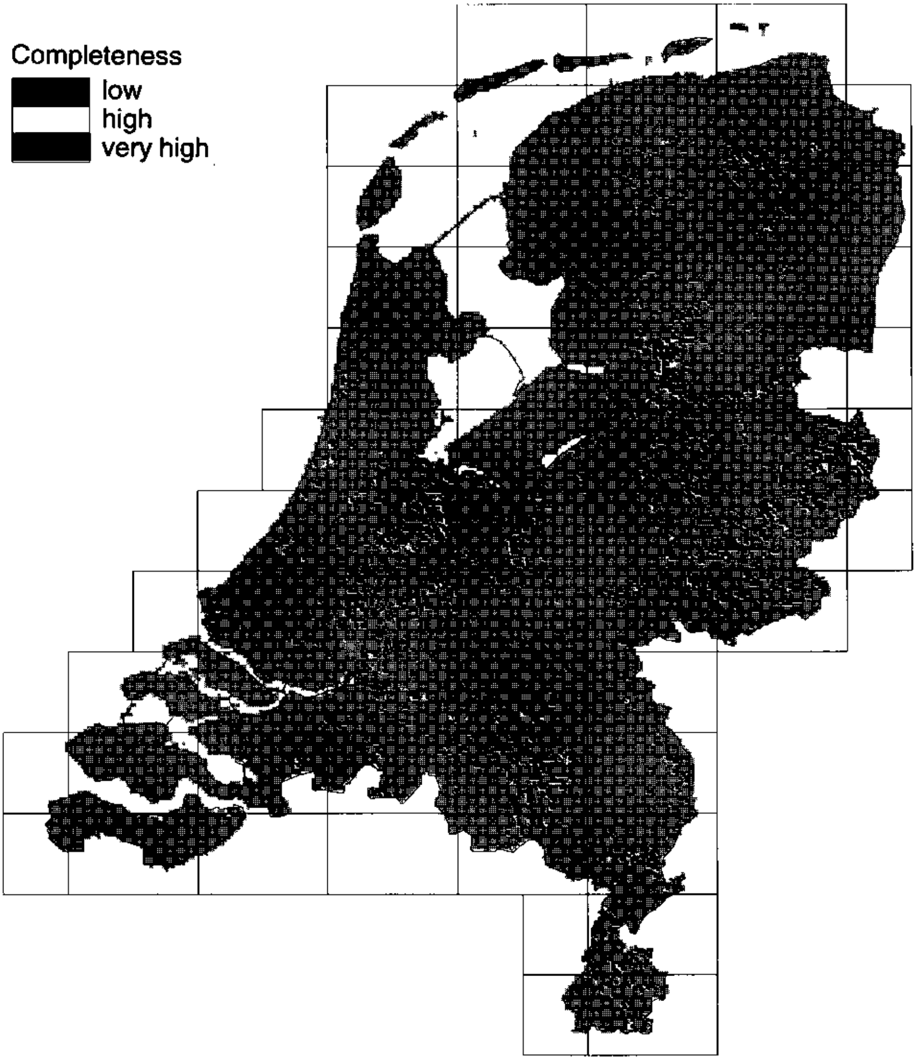


Figure 4.39: Distribution of ecotope group H27: woods and shrubs on a wet, moderately nutrient-rich soil (ecotope type H27). This ecotope group is largely bound to the valleys of brooks and rivers, and to the 'Laagveendistrict'. Based on FLORBASE-2c (gap-filled).

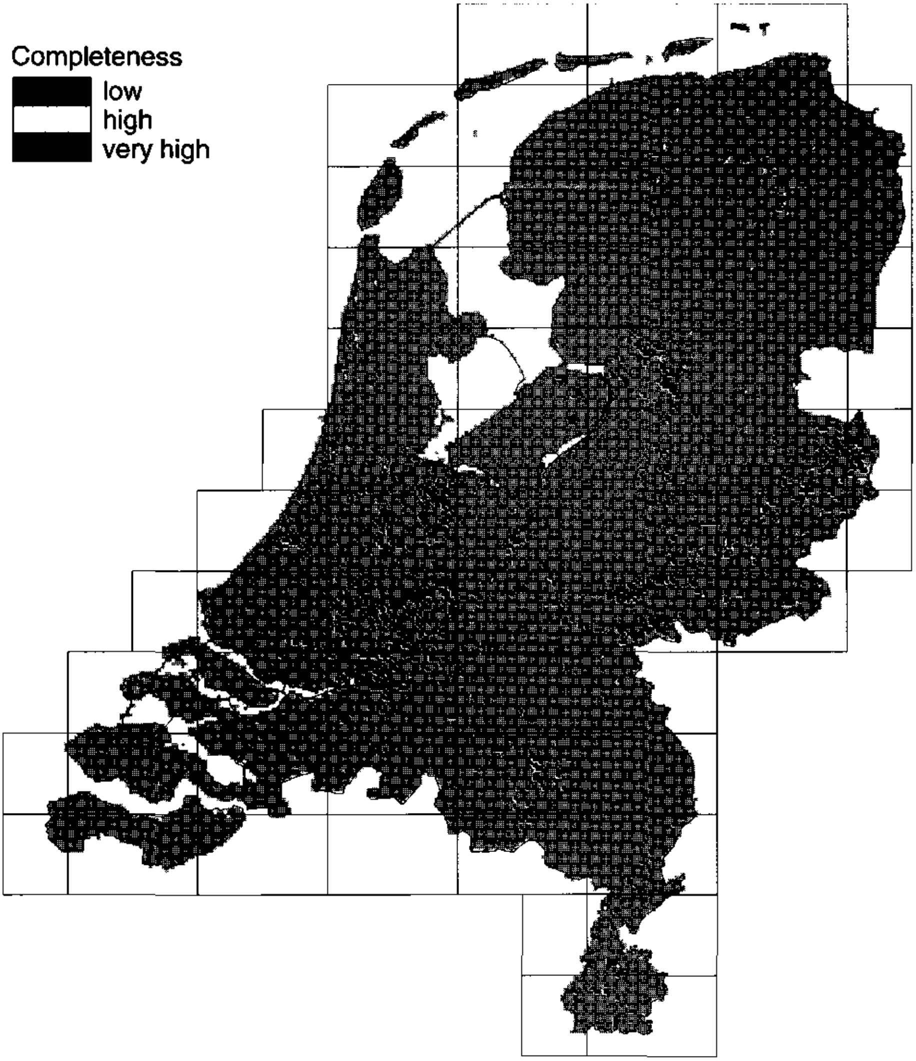


Figure 4.40: Distribution of ecotope group H28: woods and shrubs on a wet, very nutrient-rich soil (ecotope type H28). This ecotope group is especially found in the river area and in parts of the 'Laagveendistrict'. Its occurrence outside these areas may be an indication of the inlet river water or of the influence of excessive manuring. Based on FLORBASE-2c (gap-filled).

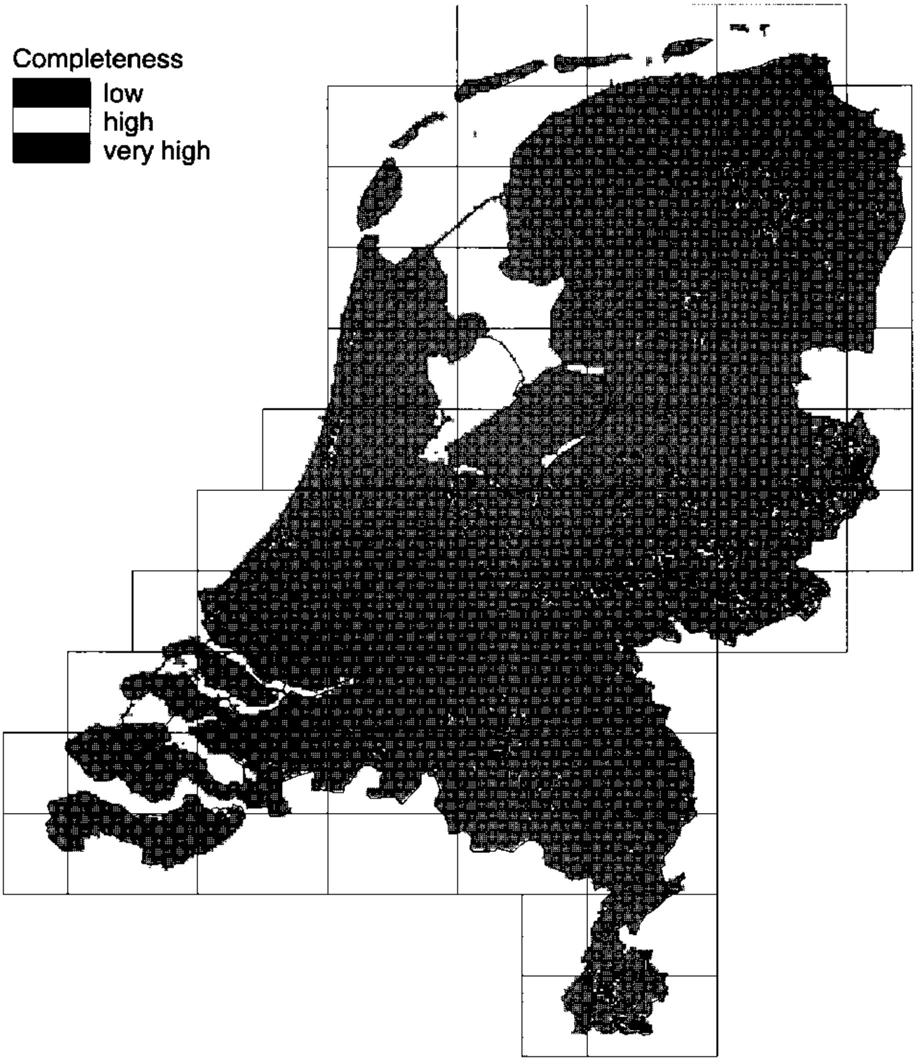


Figure 4.41: Distribution of ecotope group H42: woods and shrubs on a moist, nutrient-poor, neutral soil (ecotope type H42). This ecotope group is moderately represented in the provinces of Utrecht and Noord-Brabant. Based on FLORBASE-2c (gap-filled).

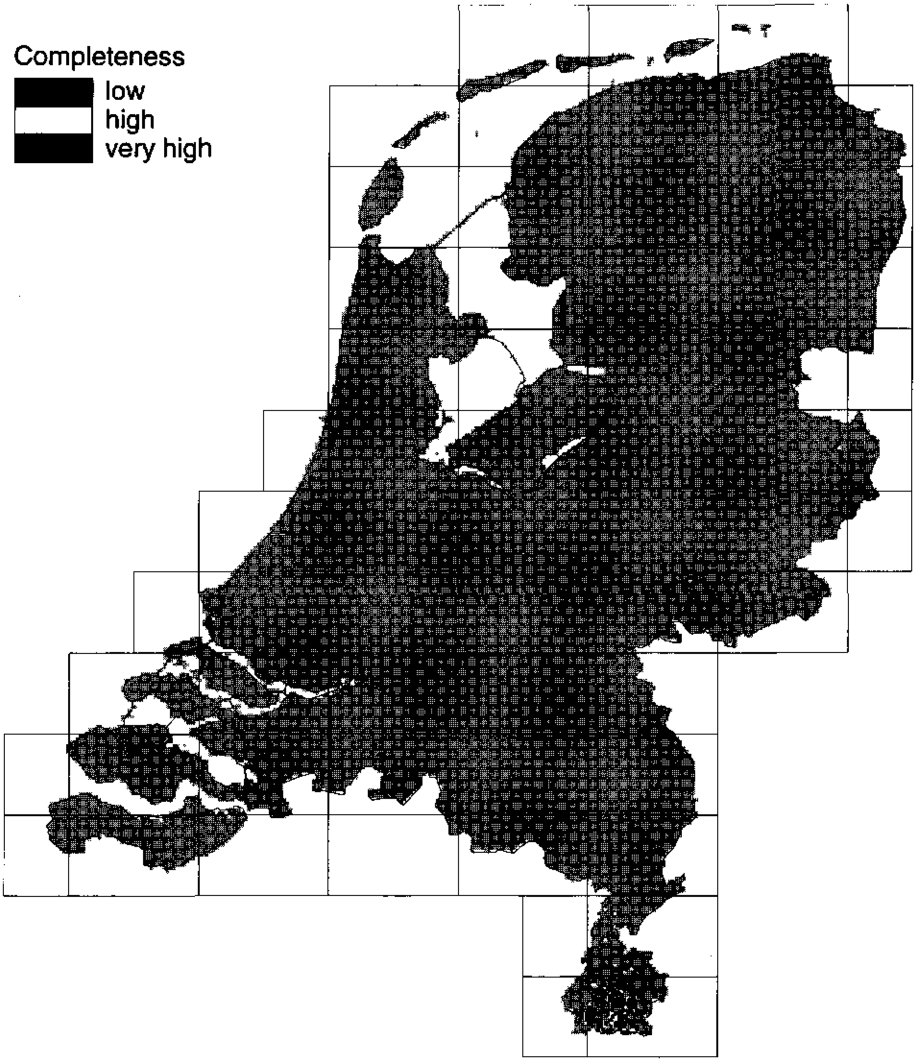


Figure 4.42: Distribution of ecotope group H43: woods and shrubs on a moist, nutrient-poor, alkaline soil (ecotope type H43). This ecotope group is almost completely restricted to the southern part of the province of Limburg. Based on FLORBASE-2c (gap-filled).

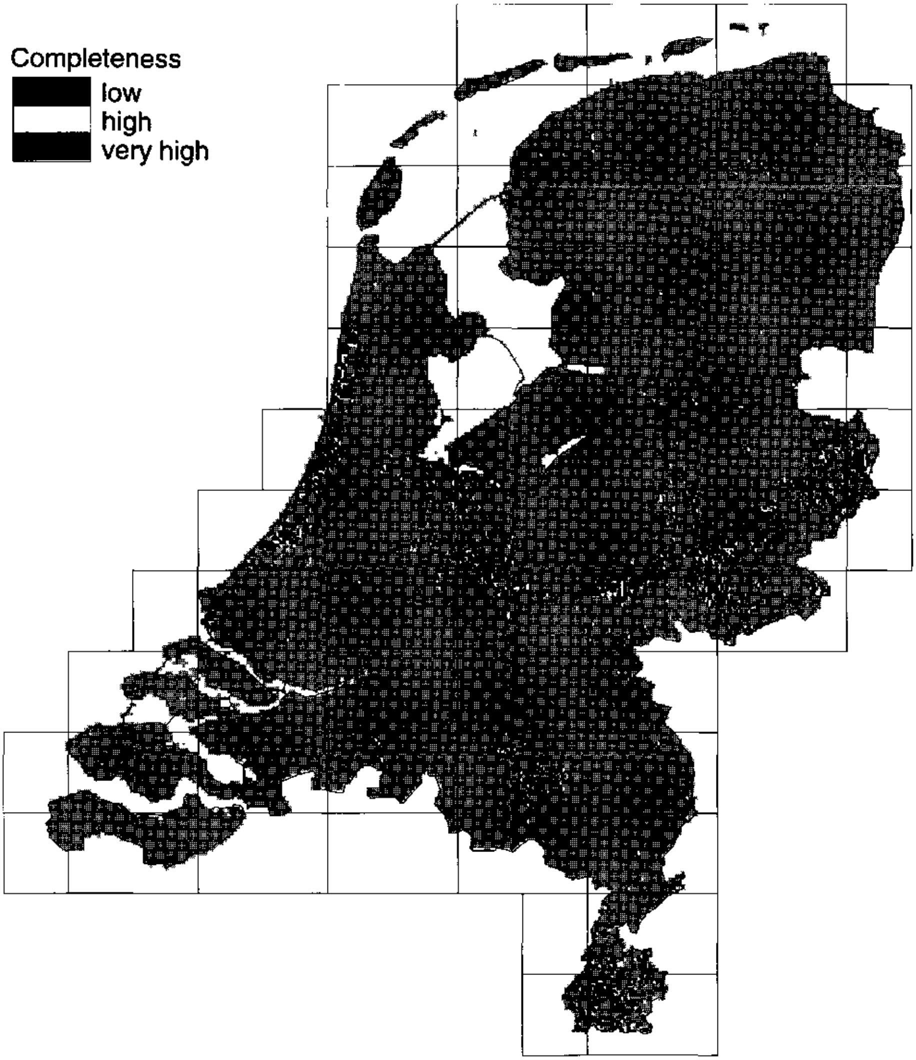


Figure 4.43: Distribution of ecotope group H47: woods and shrubs on a moist, moderately nutrient-rich soil (ecotope type H47). This ecotope group is moderately represented in the provinces of Drenthe and Noord-Brabant. Based on FLORBASE-2c (gap-filled).

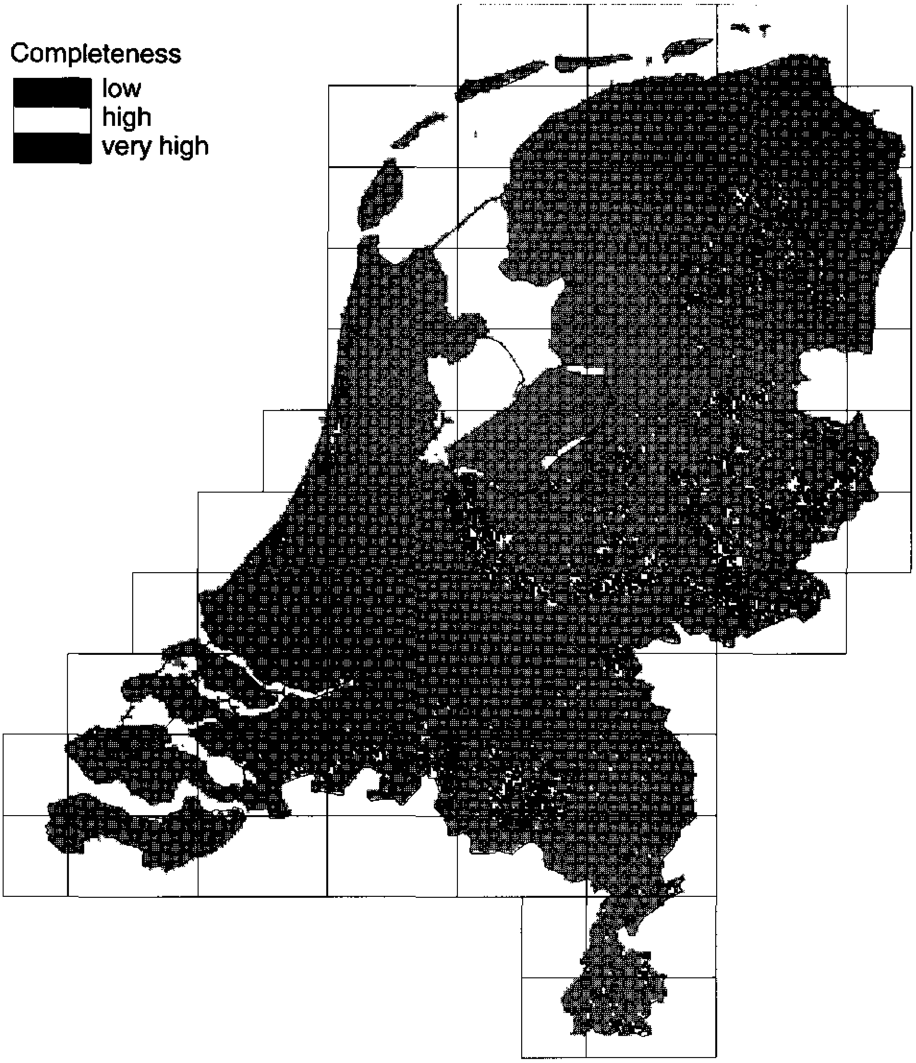


Figure 4.44: Distribution of ecotope group H62: woods and shrubs on a dry, nutrient-poor, neutral soil (ecotope type H62). As the threshold values T of this ecotope group are close to each other, the reliability of this map is low. Based on FLORBASE-2c (gap-filled).

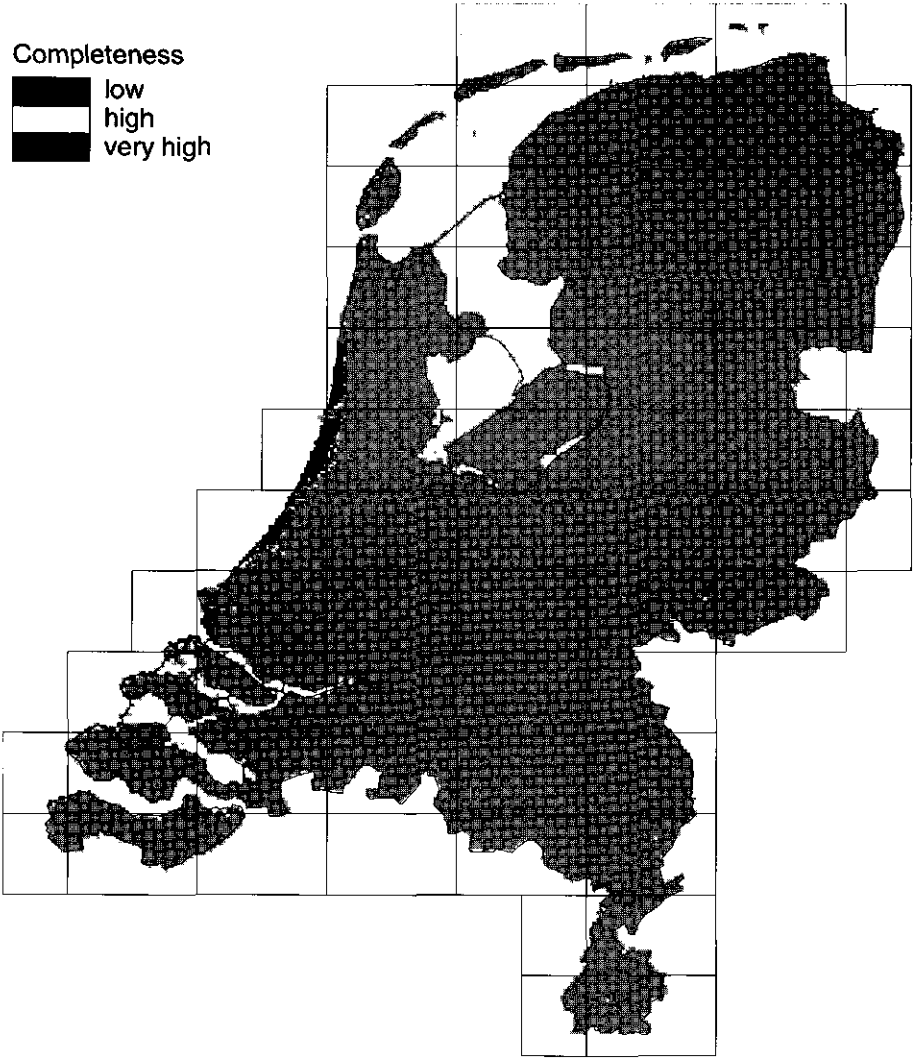


Figure 4.45: Distribution of ecotope group H63: woods and shrubs on a dry, nutrient-poor, alkaline soil (ecotope type H63). This ecotope groups is almost completely restricted to nature reserves in the calcareous dune area. Based on FLORBASE-2c (gap-filled).

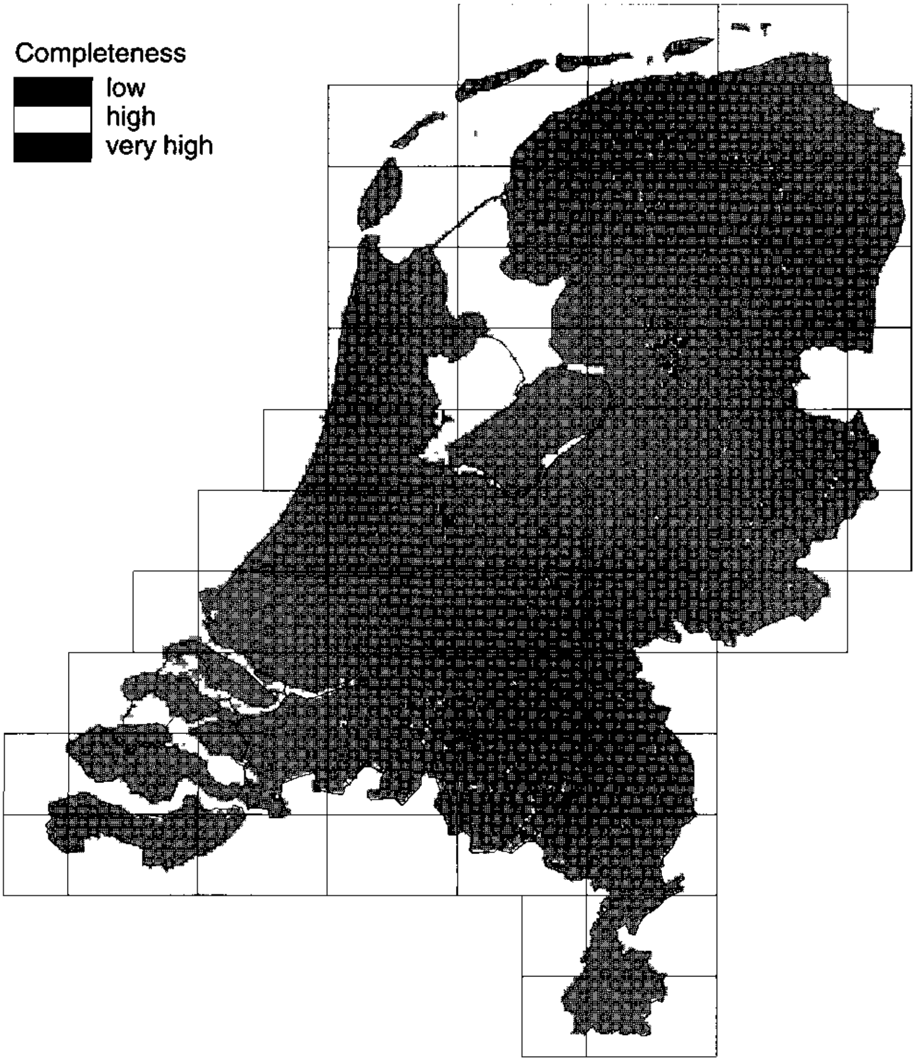


Figure 4.46: Distribution of ecotope group A12: terrestrializing vegetations and water vegetations in stagnant, nutrient-poor, neutral to alkaline waters (ecotope types V12, W12, W13). By nature, there are less indicator species on the isles of the Wadden sea than there are in the Pleistocene flora districts. It may be that the completeness on the map is too low on these isles. This ecotope group is poorly represented in the Netherlands. The map shows a picture that is probably too optimistic for the province of Noord-Brabant. Based on FLORBASE-2c (gap-filled).

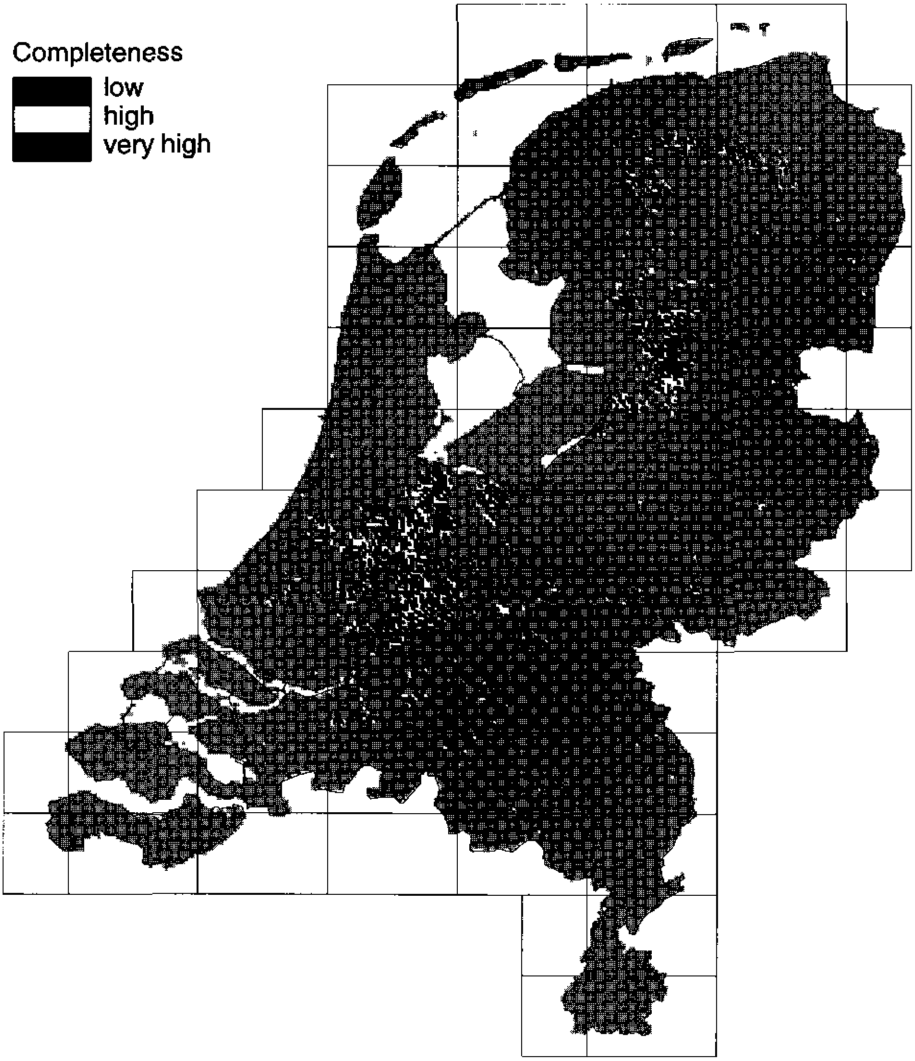


Figure 4.47: Distribution of ecotope group A17: terrestrializing vegetations and water vegetations in stagnant, moderately nutrient-rich waters (ecotope types V17, W17). This ecotope group is well-developed in the 'Laagveendistrict'. Based on FLORBASE-2c (gap-filled).

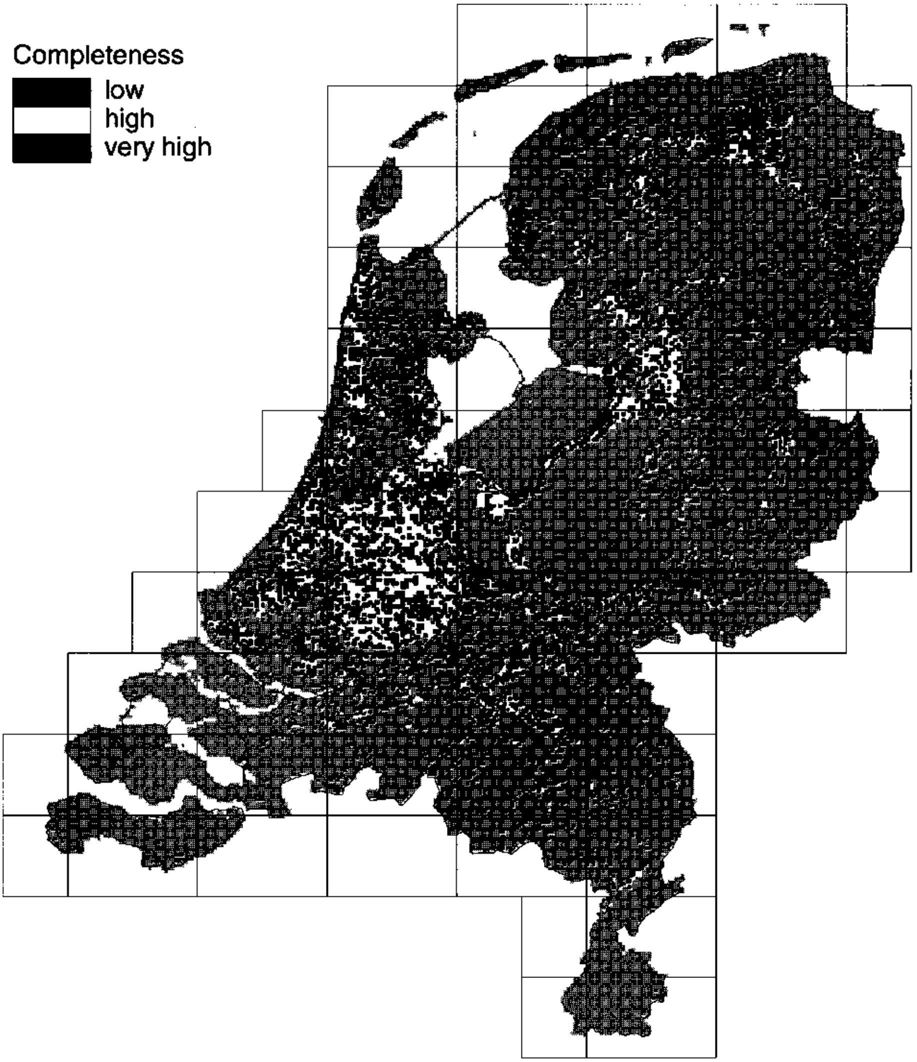


Figure 4.48: Distribution of ecotope group A18: terrestrializing vegetations and water vegetations in stagnant, very nutrient-rich waters (ecotope types V18, W18). This ecotope group is well-developed in parts of the 'Laagveendistrict'. Its occurrence outside this area may be an indication of the inlet of river water or of the influence of excessive manuring. Based on FLORBASE-2c (gap-filled).

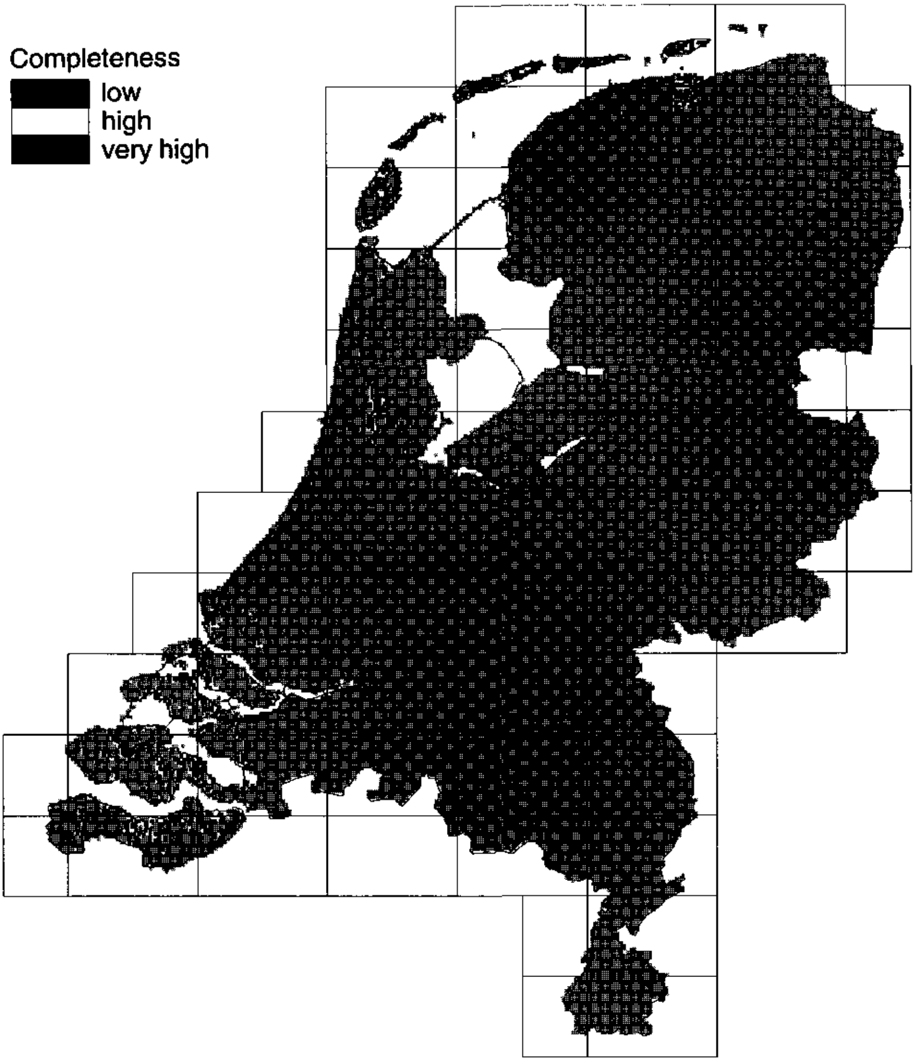


Figure 4.49: Distribution of ecotope group bK20: pioneer vegetations, grasslands and tall herbaceous vegetations on a wet, brackish soil (ecotope types bP20, bG20, bR20). This ecotope group is restricted to the 'Estuariene district', parts of the province of Noord-Holland, the west coast of the province of Friesland and the Lauwersmeer. Based on FLORBASE-2c (gap-filled).

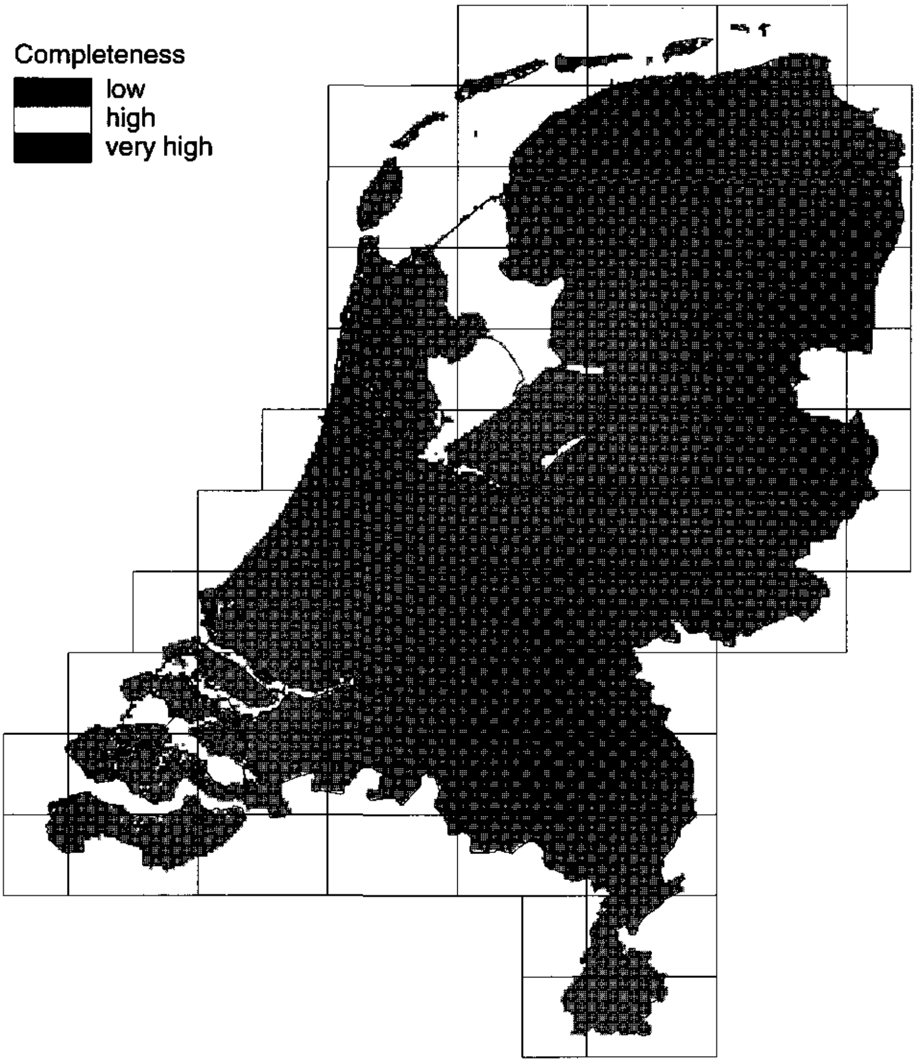


Figure 4.50: Distribution of ecotope group bK40: pioneer vegetations, grasslands and tall herbaceous vegetations on a moist, brackish soil (ecotope types bP40, bG40, bR40). This ecotope group is almost completely restricted to the 'Estuariene district', parts of the province of Noord-Holland, the west coast of the province of Friesland and the Lauwersmeer. Based on FLORBASE-2c (gap-filled).

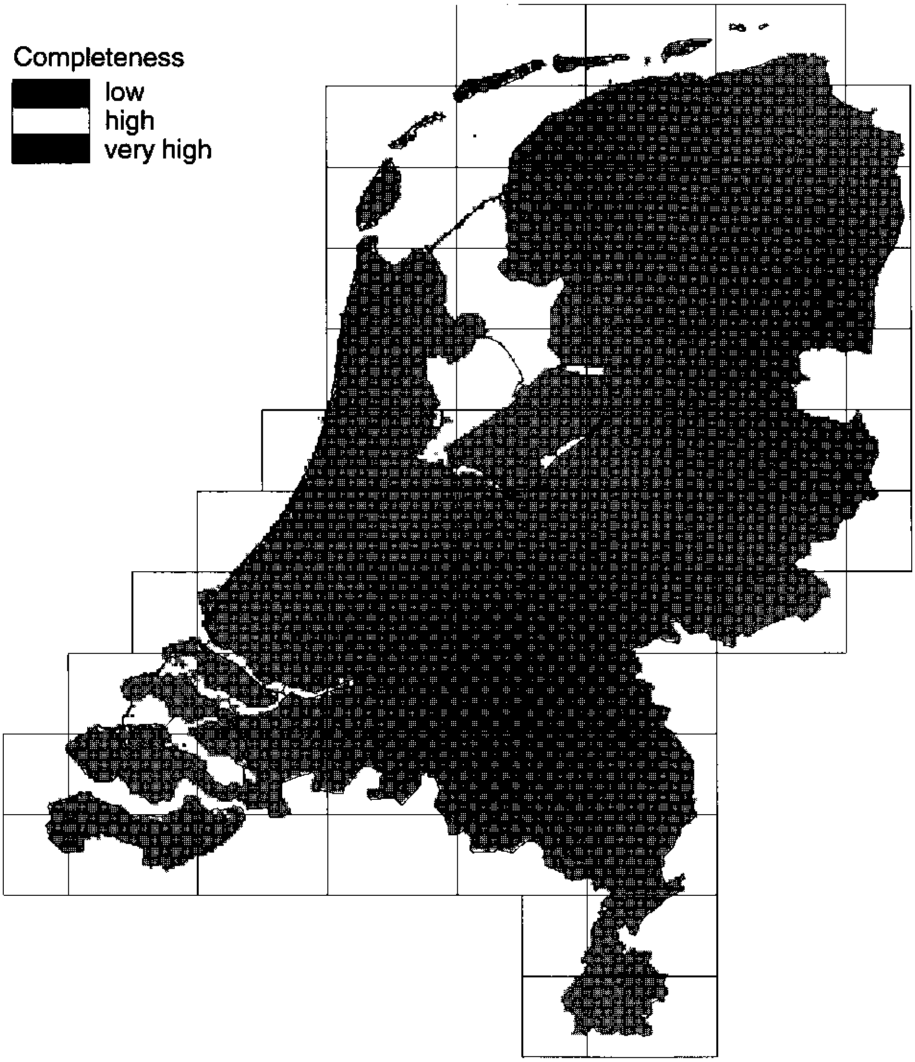


Figure 4.51: Distribution of ecotope group bK60: pioneer vegetations on a dry, brackish soil (with sand drift) (ecotope type bP60). This ecotope group is restricted to the coastal dunes. Based on FLORBASE-2c (gap-filled).

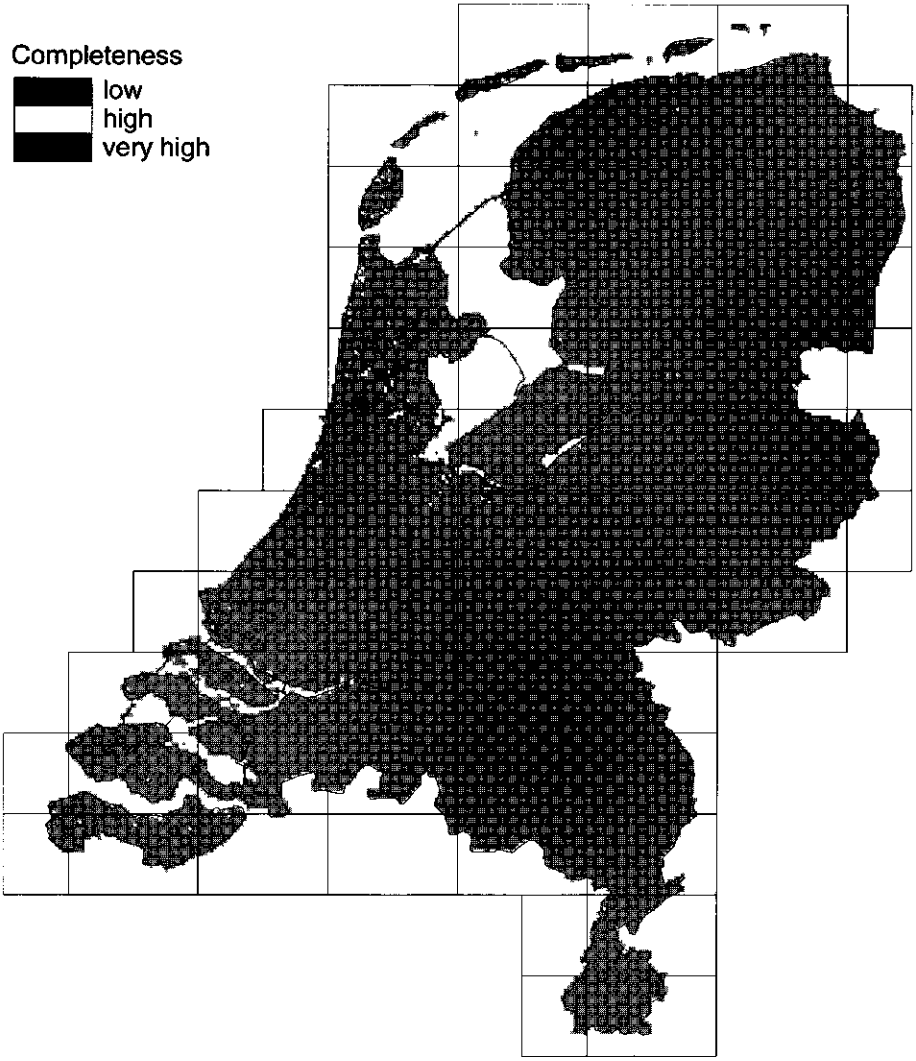


Figure 4.52: Distribution of ecotope group bA10: terrestrializing vegetations and water vegetations in stagnant, brackish waters (ecotope types bV10, bW10). The distribution of this ecotope group is concentrated in the province of Noord-Holland. Based on FLORBASE-2c (gap-filled).

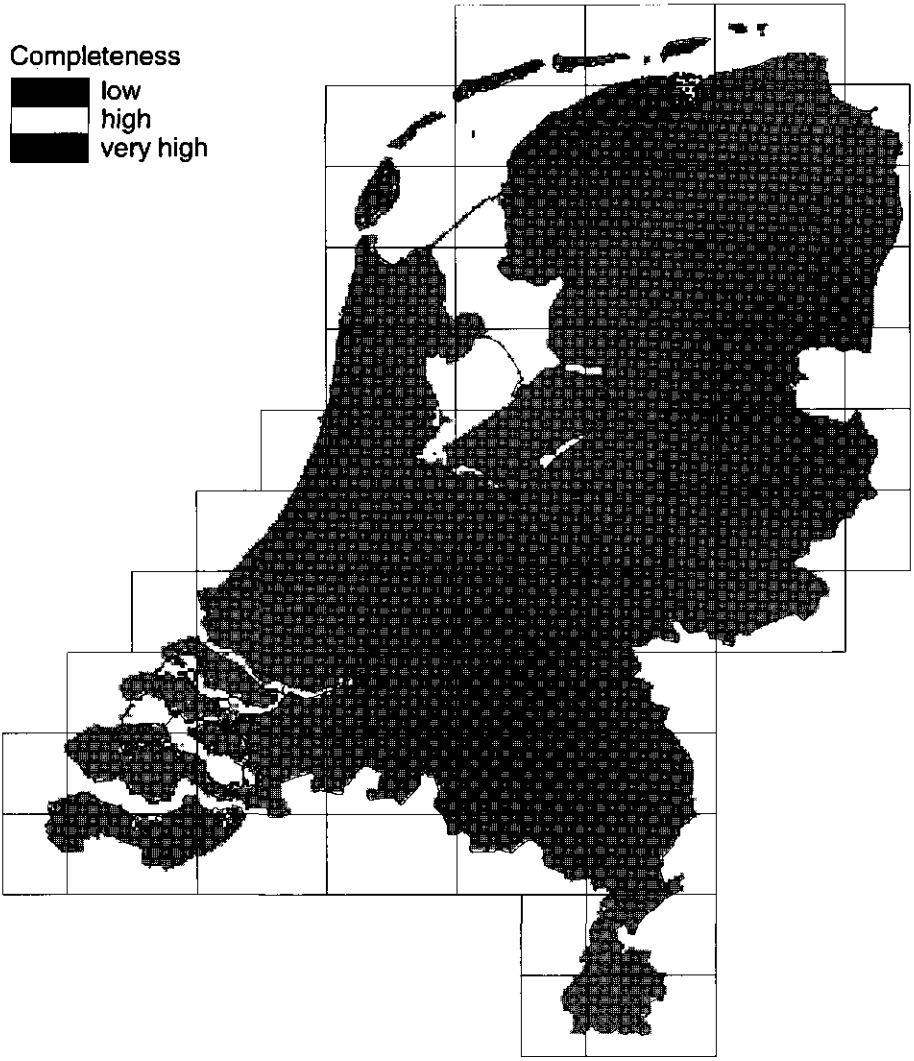


Figure 4.53: Distribution of ecotope group zK20: pioneer vegetations, grasslands and tall herbaceous vegetations on a wet, saline soil (ecotope types zP20, zG20, zR20). This ecotope group is restricted to the coast of the province of Zeeland, the Wadden sea and the Lauwersmeer. Based on FLORBASE-2c (gap-filled).

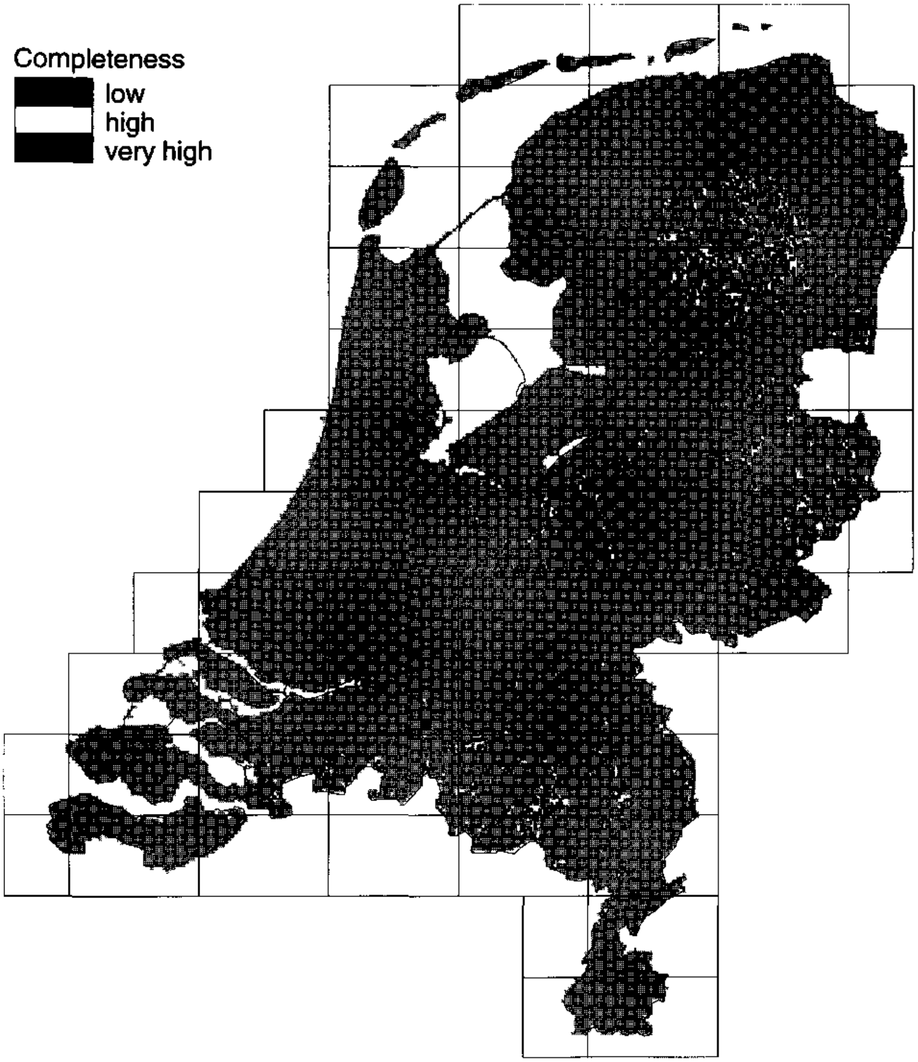


Figure 4.54: Distribution of ecotope group K21, computed without gap-filling of FLORBASE-c. Compared to the gap-filled map (Fig. 4.26), in many km-squares in the Pleistocene part of the Netherlands (especially in the province of Noord-Brabant) this ecotope group is either missing or assigned to a lower completeness class.

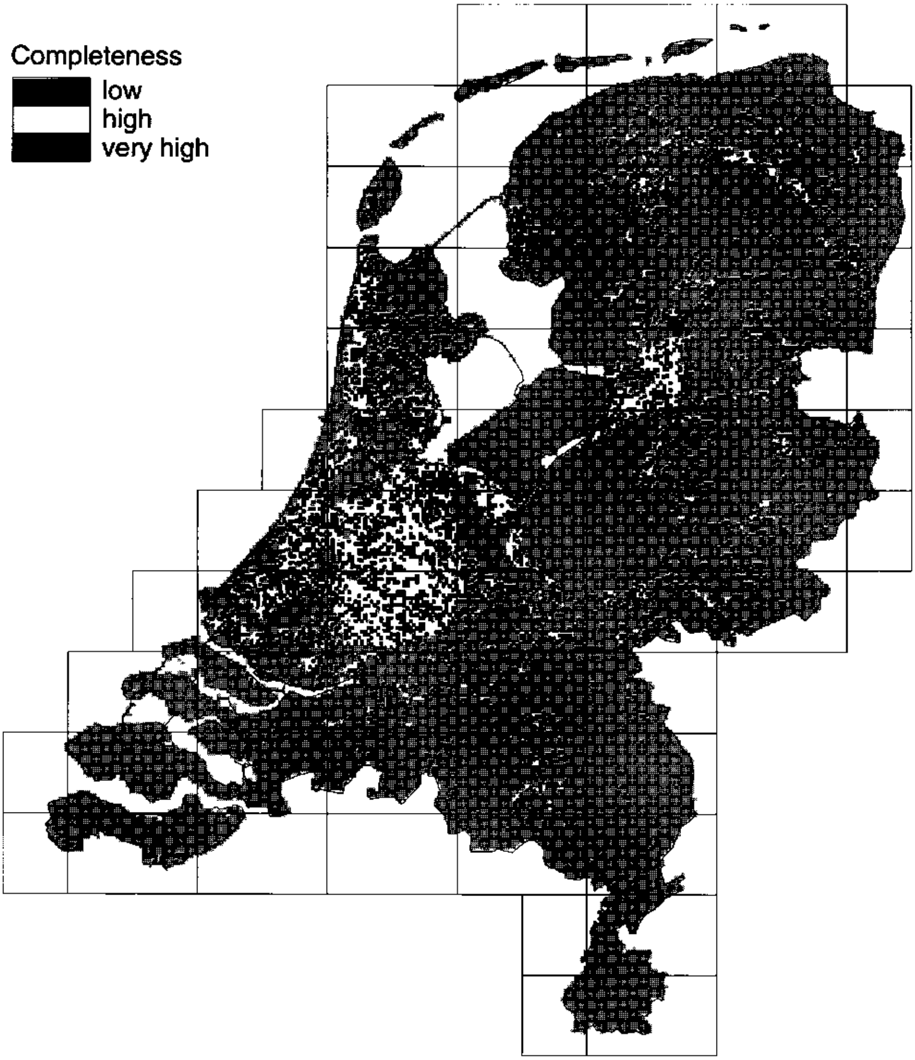


Figure 4.55: Distribution of ecotope group A18, computed without gap-filling of FLORBASE-2c. Compared to the gap-filled map (Fig. 4.48), this map lacks many km-squares in the Pleistocene part of the Netherlands, especially in the province of Noord-Brabant.

Chapter 5

The value of nature

5.1 Introduction

Any organization, pursuing an objective, needs a criterion by which the degree of realization of that objective may be measured. In most branches of our society money is the standard: the more profits and the more fortune, the better the situation. But with regard to nature conservation, money can hardly be used as a criterion, since the value of nature is not influenced by the free market mechanism of demand and supply. So what we need is an alternative criterion, which will facilitate policy-supporting research in the field of nature conservation.

In the next section (5.2) I will argue that for nature conservation valuation of *classification units* (species, vegetation types or ecosystem types), the criterion *rarity* is of decisive importance. Furthermore, several methods for measuring rarity will be discussed, with special attention for their accuracy. And finally, a mathematical formula, quantifying the conservation value of classification units on the basis of rarity, will be presented.

To valuate an actual area, not only rarity but also other criteria are important, such as species richness and the size of the area. In Section 5.3 different methods of valuation will be compared, using the national flora database FLORBASE-2c (Section 1.4.2). Subsequently, the results of the comparison will serve to select one final method for the DEMNAT model (Chapter 2).

This chapter focuses on vascular plant species, but much of what is put forward also holds for other taxonomic groups, such as birds and butterflies. Parts of this chapter have been published before by Witte (1996).

5.2 The value of rarity

5.2.1 The importance of rarity

Anyone who has ever joined a field trip of biologists knows how nature is generally valued: on the basis of rarity. During field trips it is quite common to see a biologist kneel to the ground, searching for some rare plant (Fig. 5.1). Botanists uprooting some rare plant even justify their action with the phrase: "whatever is rare should remain rare". In other words, it does not matter if one individual disappears, because the value of the remaining individuals will increase and, with that, the *total* value of the *species* remains unaltered.

In 1859 the neophyte *Elodea canadensis* (UK: Canadian Waterweed; NL: Brede waterpest) was seen in the Netherlands for the first time. It conquered all waters, rapidly becoming a pest that, according to conservationists, had to be controlled. In 1941 another member of the same genus, *E. nuttallii* (UK: Nuttall's Waterweed; NL: Smalle waterpest), popped up, multiplying just as rapidly and becoming just as unpopular as its predecessor *E. canadensis.* The latter, however, had decreased in the meantime. It has become fairly rare today and, with that, a welcome guest! This example illustrates how in a short period of time the appreciation for a certain species may drastically change. The same phenomenon, but in the reverse sense, may be observed in the case of the Blue heron (NL: Blauwe reiger): in former times rare and valuable, nowadays common and of little value.

So there is no doubt that rarity is generally used as an important criterion for the valuation of species. The same is true for the valuation of vegetation types and ecosystem types. Quagfens, bogs and wet dune slacks are much appreciated because they are rare as a type (and - consequently - harbor rare species).

Equally, much weight is ascribed to this criterion in formal methods for conservation valuation. The so-called 'red lists' of plant species that have been made for e.g. the Netherlands (Weeda *et al.*, 1990), Europe (IUCN, 1983), Belgium (Lawalrée & Devosalle, 1969), Great Britain (Perring & Farrell, 1983) and Germany (Korneck & Sukopp, 1988) are based on rarity. In 1990 the Dutch Government issued an important policy document on nature (Ministerie van Landbouw, Natuurbeheer en Visserij, 1990). The handbook "Ecosystems in the Netherlands"

Figure 5.1: Botanists, in search for rare species.

(Bal *et al.*, 1995), in which ecosystems together with their desired surface area are described, serves as a tool for reaching the objectives that are formulated in this policy document. The handbook introduces so-called *target species*, according to which the success of the policy may be measured. These species are also selected on the basis of rarity. Finally, rarity is used as a major criterion in the many quantitative procedures that serve to obtain conservation value figures (Buys, 1995; Clausman & Van Wijngaarden, 1984; Dony & Denholm, 1985; Fahner & Wiertz, 1987; Gremmen, 1986; Gremmen, 1990; Jonker & Witjes, 1994; Mennema, 1973; Stevers *et al.*, 1984; Wheeler, 1988; Witte & Van der Meijden, 1993). In some of these methods a so-called negative *trend* has also been incorporated in the calculation, indicating the extent to which the occurrence of a certain species is decreasing. Trend may be regarded as a special case of rarity: a certain species is in danger of becoming even more rare than it already is. For the time being, this criterion is omitted, but in the discussion (Section 5.4) it will be taken up again.

Why is it that we regard rare species and rare ecosystems as valuable? One motive that is often mentioned is that we feel responsible for our natural environment. With a view to its decline we feel that rare species should be cherished since they are in danger of disappearing for ever, which would reduce the biodiversity. Another motive is that rare species are ecologically interesting, for example because their rarity originates either from very specific demands of the environment or from a very specific reproduction strategy (Rabinowitz, 1981). Finally, the ordinary motive that we are collectors of rarities should not be left unmentioned.

5.2.2 The measuring of rarity

In practice, Dutch policy makers ask for information about rarity in the world, Europe or the Netherlands. Some provincial authorities even use data of rarity in their provinces (Clausman & Van Wijngaarden, 1984). This chapter focuses on rarity in relation to conservation valuation for the benefit of national policy analyses. Therefore, the rarity of a species in a relatively small area such as a municipality, is not accounted for.

The determination of the extent to which a certain species occurs in areas with the size of at least a province, requires huge inventory efforts. Consequently, it is impossible to determine rarity with great precision. In most cases a *grid* is used, allowing the occurrence of species to be indicated per cell. The first person to determine the rarity of plant species in the Netherlands in this way, was Van der Maarel (1971). He analyzed the inventory results for the inventory period 1902 – 1950, tallying for every species in how many hour squares it was found. On the basis of these numbers he proceeded by assigning these species to Hour square Frequency Classes (UFK's), according to Table 1.1 (Section 1.4.4).

By Dutch standards, one hour square of 4.17×5 km (inventory period 1902–1950) or 5×5 km (inventory period 1950 – 1980) seems sizable, but compared to other countries, this is definitely not the case. In Germany, for example, the cells of the national grid measure 11×11 km, and in England 10×10 km. In the Netherlands

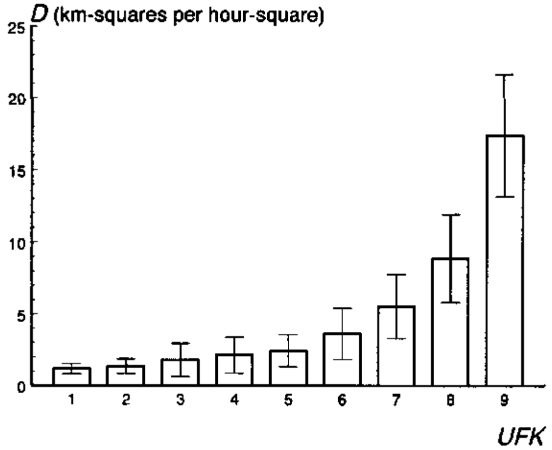


Figure 5.2: Average density D and its standard deviation per Hour square Frequency Class UFK. Density D of a certain UFK is calculated as the average number of km-squares that species with this UFK occupy within their hour squares. Example: a UFK7 species averagely occupies 5.5 km-square within its hour square. Based on FLORBASE and the UFK of species according to Van der Meijden *et al.* (1991).

nowadays even flora data per km-square (FLORBASE) exist, although rarity figures for species have not yet been derived from these. About the international rarity of plant species only rough information is available (e.g. Schaminée *et al.*, 1992).

A rarity figure that has been obtained with a grid does not reflect the actual rarity. On grid maps a species seems more common than it actually is. Fig. 5.2 illustrates that this is especially the case when rare species are concerned. It shows in how many of the 25 km-squares, species with a certain UFK are *averagely* represented within an hour square of 5 × 5 km. UFK1-species averagely cover 1.2 km-square within the hour square in which they occur. On the other hand, UFK9-species have an average *density* D of 17.4 km-squares within their hour square (see also column D in Table 5.2, Section 5.2.4). Not only is a species with a low UFK rare in the Netherlands on the level of hour squares, *it is also relatively rare within the hour squares in which it occurs.*

For many applications it does not matter if species seem more common than they actually are, as long as the *order* of rarity of the species is determined in the proper way. However, Fig. 5.3 shows that *this order may be influenced by the size of the cells.* In the figure the *relative occurrence* in the Netherlands is shown for two UFK5-species (UFK according to Van der Meijden *et al.*, 1991), at variable cell-widths (square cells), 'relative occurrence' meaning the percentage of cells, with which the species in question covers the Netherlands. Up to a cell width of 13 km *Apium inundatum* (UK: Lesser Marshwort; NL: Ondergedoken moerasscherm) appears to be the rarest species, but as soon as the cells become larger

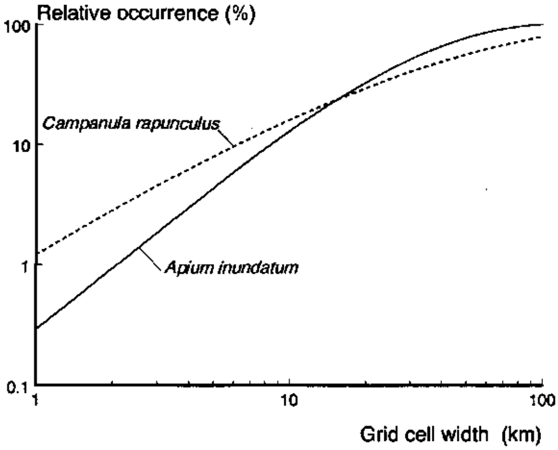


Figure 5.3: Relative occurrence of *Apium inundatum* and *Campanula rapunculus* in relation to grid-cell width (square cells) (based on FLORBASE). The larger the cell, the more common a species seems to be. For large cells - in this particular case > 13 km - species with a diffuse distribution pattern (*Apium inundatum*) are estimated to be more common than species that have a clustered distribution pattern (*Campanula rapunculus*).

Campanula rapunculus (UK: Rampion Bellflower; NL: Rapunzelklokje) beats the lot. This ordershift has to do with the difference in distribution pattern: *Apium inundatum* is diffusely represented in all of the Netherlands, whereas *Campanula rapunculus* has a clustered distribution pattern, with a concentration along the river Meuse in the province of Limburg.

One might wish to be informed about the actual rarity of species and ecosystems, but whether this is possible should be seriously questioned. How should for instance the numbers of specimens be counted of species that densely cover the ground? Or of species that multiply by means of rootstocks? What to do with the tiny seedlings and with the numerous plant individuals hiding in the soil in the form of seeds? And if we want to determine the extent to which a species covers the ground, at what time during the growing season should that be done? Many forms of rarity have to be seen in relation with the plant's way of growing (Barkman, 1968; Rabinowitz, 1981), how do we deal with that? There are for example rare species that locally abound in colonies, whereas other species occur as solitary growers. Take for instance two species with an identical *UFK*, a flimsy colony-forming species (*Rhynchospora fusca*; UK: Brown Beak-segde; NL: Bruine snavelbies) and a sturdy, solitary species (*Angelica archangelica*; UK: Garden Angelica; NL: Grote engelwortel). Should the colony-former be regarded as more rare since its total cover of the Netherlands is smaller, or should it be regarded as more common since the total number of individuals is higher? Finally, there is the problem of time-related rarity (Barkman, 1968): species may for instance be temporarily common, whereas they are usually rare.

For these as well as for numerous other problems, it is impossible to give solutions that are both practical and fundamental. *Actual rarity does not reveal itself.* At best, agreements may be reached as to how to 'solve' problems in a technical way. All in all, we must conclude that rarity can only be determined with limited precision.

5.2.3 A mathematical formula for valuing on the basis of rarity

For computation ends it may be necessary to convert occurrence measures (or rarity measures) into numerical conservation value figures on a *cardinal* scale. In this section I will present a general applicable mathematical formula that can be used for this. With the symbol A I will indicate the degree of occurrence by which a certain classification unit is represented in a certain area (for example in the Netherlands). A may be expressed in various ways, for instance in hour squares or hectares. For the conservation value I will use the symbol V.

All current quantitative valuation procedures are based on the following assumption:

Assumption 1: *The more rare, the more valuable*

This assumption is comparable with the economic law of the price elasticity of supply, stating that the price of a product rises as its scarcity increases. A graphical translation is given in Fig. 5.4. The classification unit (species/type) that is the most rare ($A = A_{\min}$) has the highest value ($V = V_{\max}$); the classification unit that is the most common ($A = A_{\max}$) the lowest ($V = V_{\min}$).

It is an assumption of course, to which exceptions exist and for which conditions may be mentioned. One condition is for instance that species should not be a nuisance: A rare butterfly is nice, a tapeworm (NL: Lintworm) is not. Another condition is that the assumption should only be applied to wild species. A rare weed species in a corn field (*Legousia speculum-veneris*; UK: Large Venus's-looking-glass; NL: Groot spiegelklokje) has much conservation value, whereas the same species planted in a garden has none. One more condition is that the assumption can only be used to compare species of the same major taxonomic groupings (mammals, birds, vascular plants, etc.). It is not practical to compare for instance a Slipper animalcule (NL: Pantoffeldiertje) with a Spoonbill (NL: Lepelaar), since single-celled organisms do not appeal to us the way birds do.

The question now arises what the function $V = \mathrm{f}(A)$ should look like. The left part of Fig. 5.5 offers some possibilities. In Fig. 5.5A, V is inversely proportional to A: if, for instance, one species is twice as rare as another, its value is twice as high. This method is used in many valuation procedures, for instance in the ecohydrological prediction model WAFLO (Section 2.2; Table 5.1). A logarithmic function is shown in Fig. 5.5B. The UFK's of plant species that are often used for conservation valuation are based on a (semi-)logarithmic scale (e.g. Gremmen, 1986). As far as I know, there is, as yet, no procedure where a linear relationship (Fig. 5.5C) is applied.

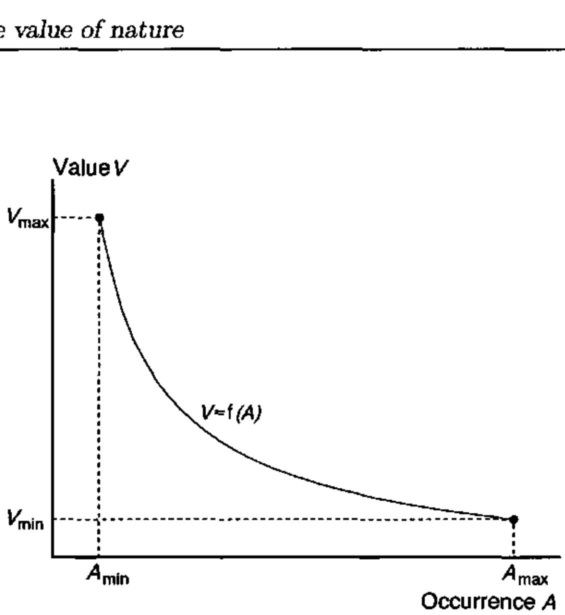
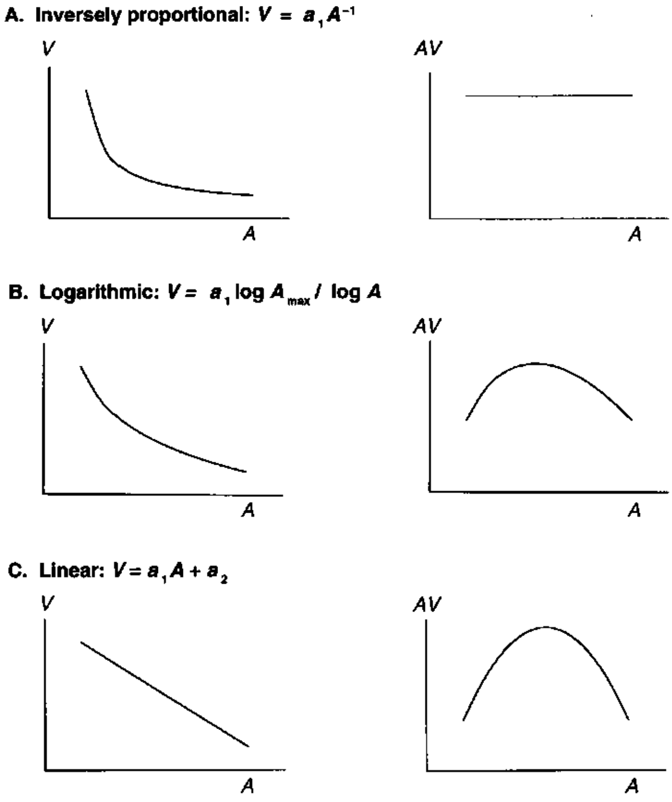


Figure 5.4: Assumption 1 used for the conservation valuation of classification units (species, vegetation types, ecosystem types): the more rare a unit is (low occurrence A), the higher its value V.

Table 5.1: Conservation values V per UFK-class, as used in the WAFLO model (Reijnen *et al.*, 1981). According to the WAFLO method, the value of a species is inversely proportional to its occurrence A. As a measure of A, WAFLO uses the average number of hour squares per UFK. Van der Maarel (1971) deduced these averages from distribution data of separate plant species of the inventory period 1902 – 1950 (hour squares of 5 × 4.17 km).

UFK	Average number of hour squares (5×4.17 km)	Conservation value V
1	2	712.
2	6	237.
3	18	79.
4	48	30.
5	121	12.
6	279	5.1
7	540	2.6
8	927	1.5
9	1423	1.0



A. Inversely proportional: $V = a_1 A^{-1}$

B. Logarithmic: $V = a_1 \log A_{max} / \log A$

C. Linear: $V = a_1 A + a_2$

Figure 5.5: Relation between occurrence A and conservation value V (left graphs), as well as totalized conservation value AV (right graphs), according to several conservation valuation functions $V = \mathrm{f}(A)$. The graphs on the left illustrate the general applied assumption that the value V of a classification unit (species, vegetation type, ecosystem type) increases as its occurrence A decreases. The right-hand graphs show that the totalized value $A \times V$ strongly depends on the kind of function $V = \mathrm{f}(A)$.

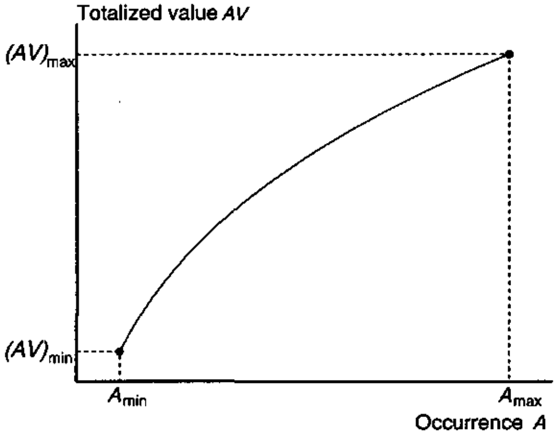


Figure 5.6: Assumption 2 used for the conservation valuation of classification units (species, vegetation types, ecosystem types): the more common a classification unit becomes (increasing occurrence A), the higher its totalized value AV.

I propose to determine the form of $V = \mathrm{f}(A)$ with the following assumption:

Assumption 2: *The more, the better*

With this I mean that an increase of A should yield a positive judgement. After all, nature policy is geared towards 'producing' as much nature as possible. A measure for the total amount of conservation value of a classification unit is the product of its occurrence A with its value V, resulting in the *totalized value* AV. In order to meet Assumption 2, $V = \mathrm{f}(A)$ has to be chosen in such a way that AV increases as A increases. A graphical translation of Assumption 2 is given in Fig. 5.6.

In the right part of Fig. 5.5, A is plotted against AV for different methods. The WAFLO-method (Fig. 5.5A) results in a horizontal line: whatever measures are taken, the totalized conservation value AV of the Netherlands never increases. (Consider for example a species with an UFK of 3. According to Table 5.1 it is assigned a occurrence A of 18 hour squares to by the WAFLO model, and a conservation value V of 79. Suppose now that many years after implementation of a certain nature-minded policy this species gets a UFK of 6 (with $A = 279$). Then, according to the WAFLO method, its value has dropped to $V = 5.1$, leaving its totalized value AV intact: $18 \times 79 \approx 279 \times 5.1 \approx 1420$). Figs. 5.5B and 5.5C are peculiar: above a certain rate of A it appears that the more nature thrives, the lower its totalized value becomes. We would have to fight certain species in order to increase their totalized value!

I propose the following formula for $V = \mathrm{f}(A)$ because it meets Assumption 1 as well as Assumption 2 (Witte & Van der Meijden, 1993):

$$V = V_{\min}\left(\frac{A_{\max}}{A}\right)^{c} \tag{5.1}$$

$$\text{where:}\quad c = \log\left(\frac{V_{\max}}{V_{\min}}\right)\left[\log\left(\frac{A_{\max}}{A_{\min}}\right)\right]^{-1}$$

$$\text{with the condition:}\qquad \frac{A_{\min}}{A_{\max}} < \frac{V_{\min}}{V_{\max}} < 1$$

Appendix E describes how Eq. (5.1) was mathematically derived.

Eq. (5.1) may be specified by linking to $A_{\max}$ the value $V_{\min} = 1$, which is the starting point of the V-scale. Subsequently, the length of the scale $V_{\max}$ can be based on Helliwell (1973), who claims that the totalized value of a species decreases by 50% when the number of its individuals diminishes by 85%. This results in:

$$0.5AV = 0.15A \times V(0.15A) \tag{5.2}$$

To come up to this statement, Eq. (5.1) has to be equipped with an exponent c of 0.63 (see Appendix E), leading to:

$$V = \left(\frac{A_{\max}}{A}\right)^{0.63} \tag{5.3}$$

5.2.4 Application of the valuation formula

Valuation of species

Information about the current UFK of species is presented in the 'Standard list of the Dutch flora', which is published every few years (see Section 1.4.4). This information may serve to determine the occurrence A of species in the Netherlands and, subsequently, their conservation value V with Eq. (5.3).

In Table 5.2 for each UFK the corresponding average number of 5 × 5 km hours squares M is given. M deviates from the figures shown in Table 5.1, since Van der Maarel (1971) calculated the latter from distribution data of the inventory period 1902 – 1950. As the hour squares of this inventory period measure 5 × 4.17 km in stead of the current size of 5 × 5 km, I multplied his figures with $\frac{4.17}{5} = 0.83$ to obtained a rough estimation of M.

M may serve as a measure of occurrence A. From Table 5.2 it appears that $A_{\max}$ amounts to 1186, representing the M-value belonging to UFK9. When Eq. (5.3) is applied to this rate and the M-values, we obtain the conservation values presented in the forth column of Table 5.2.

Table 5.2: Conservation values V per UFK, computed with Eq. (5.3). The values on the 1 : 31-scale were obtained by using the average number of hour squares (of 5×5 km) per UFK (M) as a measure of occurrence A ($A = M$, $A_{\min} = 5$, $A_{\max} = 1186$). In order to get values that are applicable to km-squares, A was also computed by multiplying M with density D, i.e. with the average number of km-squares per UFK. This resulted in the V-values on the 1 : 153-scale ($A = MD$, $A_{\min} = 5 \times 1.4$, $A_{\max} = 1186 \times 17.4$).

UFK	M	D	V	
			1:31	1:153
1	2	1.2	31.	153.
2	5	1.4	31.	153.
3	15	1.8	16.	66.
4	40	2.1	8.5	32.
5	101	2.4	4.7	16.
6	233	3.6	2.8	7.5
7	450	5.5	1.8	3.8
8	773	8.9	1.3	2.0
9	1186	17.4	1.0	1.0

At UFK1 a minimum M of 2 is read from Table 5.2, but this rate has not been used as $A_{\min}$, since the UFK in which extremely rare species are put, highly depends on the inventory efforts. Another reason why this rate is not used as $A_{\min}$ is that there is an element of dynamics and chance in the appearing and disappearing of rare species. Additionally, extremely rare species may be nothing but 'curiosities', to which - according to Helliwell (1973) - no high value should be ascribed. To compensate for this, a minimal rate of 5 is used for $A_{\min}$, so that the maximum conservation value calculated with Eq. (5.3) amounts to $V_{\max} = V(5) = 31$.

We may feel that a 1 : 31-scale is a bit short, but this is because we are not used to judging on the basis of hour squares. This becomes clear when we consider the totalized value AV: for example 340 at UFK4 (40×8.5) and 1005 at UFK8 (773×1.3). So when a UFK4-species extends to a degree that it can be classified in UFK8, its total contribution to the conservation value of the Netherlands is raised by a factor 3. This may seem very little, but for a larger factor we would have to choose a conservation value-scale that is even shorter than this one! According to the condition of Eq. (5.1) the value-scale should in all cases fall within $A_{\min} : A_{\max} = 1 : 237$. In other words: *it should be shorter than the scale that is implied by the inventory data.*

In the above argument the hour square is taken as a unit of application. In practice, it is often necessary to valuate smaller units, such as vegetation relevés or km-squares. A 1 : 31-scale would definitely be too short in these cases, since it would lead to an over-valuation of common species. For these kind of applications it is necessary to stretch the scale for occurrence-rates. In general, this can be done by multiplying occurrence figures from a grid with the species average density within the grid-cells (see Clausman & Van Wijngaarden, 1984). Unfortunately, average density data are not available for the Netherlands. But for conservation valuation with FLORBASE we may rely on UFK-figures, that is, by multiplying the average number of hour squares per UFK - M - with the average number of km-squares per hour square - D. In this simple, albeit somewhat rough way, an estimation is obtained of the total amount of km-squares covered by a certain UFK-species in the Netherlands. According to this method, a UFK1-species averagely occurs in $A = 2 \times 1.2 = 2.4$ km-squares, and a UFK9-species in $A = 1186 \times 17.4 \approx 21,000$ km-squares. Conservation values that are based on this occurrence measure, as well as on Eq. (5.3), are also incorporated in Table 5.2 (last column). These values happen to resemble the WAFLO-values (Table 5.1), that have been used in practice for vegetation relevés.

Valuation of ecosystem types

In many valuation methods the conservation value of a spatial entity - such as a vegetation relevé or a kilometer square - is obtained by adding up the conservation values of the occurring species. The resulting rate reflects the diversity of the species concerned, as well as their conservation values. An objection against this approach is that certain highly valued but nevertheless species-poor ecosystems (bogs, salt marshes, drifting sand dunes, heathlands) get too low a valuation. Another drawback is that *all* species add to the total value in a positive sense, including the species that are part of disturbances. In that case, a heathland scores higher when it contains weed-covered garbage.

The national distribution maps about the botanical quality (completeness) of different ecotope groups (i.e. combinations of ecotope types of the same site), drawn by Witte & Van der Meijden (1995) (Chapter 4, Figs. 4.26-4.53), partly make up for these objections. For computation ends, the completeness shown on these 'ecotope maps' can be expressed by a variable C, varying from 0 (for 'noise') to 1 (for 'very high'), see Section 4.4.3. Summation of C over all the n kilometer

Table 5.3: Occurrence values A of ecotope groups, plus conservation values V derived from A with Eq. (5.3). The conservation values of the asterisk-marked ecotope groups have been raised by 20%, since they contain many species that are internationally rare. See Table 4.2 for a description of the ecotope groups and the ecotope types they are derived from.

	A	V		A	V		A	V		A	V
*K21	700	3.8	K43	280	5.7	H28	1310	2.1	A17	1180	2.3
*K22	600	4.2	K46	650	3.3	H42	780	2.9	A18	4250	1.0
*K23	230	7.7	K61	850	2.8	H43	120	9.7	bK20	560	3.6
K27	1780	1.7	K62	1700	1.8	H47	1230	2.2	bK40	530	3.7
K28	3950	1.0	K63	360	4.8	H62	1160	2.3	bK60	270	5.7
*K41	820	3.4	*H22	250	7.2	H63	400	4.5	bA10	350	4.9
K42	640	3.3	H27	1760	1.8	*A12	160	9.8	*zK20	500	4.7

squares of the Netherlands yield an occurrence measure A of the ecotope group concerned in the Netherlands: $A = \sum_{i=1}^{n} C_i$. A-values computed in this way are given in Table 5.3. So in this A-measure also the quality of ecotope group is incorporated: an ecotope group becomes more common as its portion of fully developed km-squares rises. Conservation values are derived from the occurrence figures with Eq. (5.3), see Table 5.3. Some of the V-values have been slightly raised, since they harbor many species that are internationally rare according to Schaminée *et al.* (1992). See Witte & Van der Meijden (1993) for more information about the calculation method. The figures of Table 5.3 should be considered as *potential conservation values*, since they are ascribed to botanically very well-developed km-squares (completeness class 'very high') of the ecotope group in question.

5.3 Botanical valuation of kilometer squares

5.3.1 Method and results

For provincial and national conservation policy there is often no other choice but to rely on the system FLORBASE, which provides observations per km-square. FLORBASE is for instance used in the national models MOVE (Section 2.2) and DEMNAT (Section 2.3), as well as in provincial and even regional valuation methods (e.g.: Groen, 1997; Jonker & Witjes, 1994; Runhaar & Groen, 1993). We will now examine how FLORBASE can be used for the botanical valuation of km-squares. For this we will compare eight methods. The comparison focuses on the province of Utrecht (Fig. 2.15), since the inventory of this area was carried out thoroughly. For each method, a color map has been made, showing the botanical values in four classes: 'very low', 'low', 'high' and 'very high' (Figs. 5.7-5.10). The borderlines between the classes have been chosen in such a way that the distribution of the number of km-squares among the classes is approximately the same for every method. Between the methods, also Spearman's rank correlations with ties (Sachs, 1982, p. 401) - r_S - have been calculated (Table 5.4).

The first 5 methods that I will describe below are based on *species diversity*, as well as on *conservation values for species*. The last three methods are based on the *ecotope maps* of Chapter 4 (Figs. 4.26-4.53). These ecotope maps can be put together in numerous ways, forming a conservation value map. There may be fluctuations in for example the *length of the conservation value scale*, as well as in the degree to which the result is determined by the *diversity of ecotope groups* within a km-square. I will deal with three out of the twenty possibilities that I have examined.

i. Species richness
According to most valuation methods the criterion *diversity* is important for the valuation of areas (Margules & Usher, 1981). The simplest measure for species-diversity is species richness. Fig. 5.7 gives the conservation value-map, which is based on the number of species per km-square.

ii. Sum conservation values of species (1:31)
Some species should receive a higher value than others, for instance because they are rare on a national level. This method is based on the sum of the conservation values of species on the 1 : 31-scale of Table 5.2. No map is presented of this method.

iii. Sum conservation value of species (1:153)
Conform method ii, but this time with the 1 : 153-scale of Table 5.2 (no map).

iv. WAFLO
As method ii, but this time with the values of the WAFLO-model (Table 5.1). See Fig. 5.8

v. Number of target-species
The handbook "Ecosystems in the Netherlands" (Bal *et al.*, 1995) - see Section 5.2.1 - contains a list of target species meant for conservation policy. These species meet at least 2 out of the following 3 criteria: (1) in international respect the Netherlands form an important area for the species in question (i-criterion), (2) the species is rare in the Netherlands (z-criterion), (3) the species shows a negative trend in the Netherlands (t-criterion). The number of target species per km-square forms the fifth valuation method. See Fig. 5.9.

vi. Sum completeness of ecotope groups
The simplest way to combine the ecotope maps is adding up the completeness fractions C of all ecotope groups within a km-square ($V_{\mathrm{km}} = \sum_{e=1}^{n_e} C_e$, where V_{km} = conservation value km-square, C = completeness fraction, e = ecotope group, n_e = total number of ecotope groups, i.e. 28). Hence, the resulting figure is based on the relative diversity of species per ecotope group, as well as on the diversity of ecotope groups within a kilometer square (no map).

vii. Sum conservation values of ecotope groups
As is the case with species, some ecotope groups may be given a higher value than others. This is done by multiplying the completeness fraction with the conservation value according to Table 5.3, upon which the results are added up ($V_{\mathrm{km}} = \sum_{e=1}^{n_e} C_e V_e$, where V_e = conservation value ecotope group e). See Fig. 5.10.

viii. DEMNAT-2.0
Witte & Van der Meijden (1993) also tried to incorporate the *size* of the ecotope group within the km-square in the valuation. Their method - used in DEMNAT-2.0 - makes use of a weight-factor for this size ($V_{\mathrm{km}} = \sum_{e=1}^{n_e} W_e C_e V_e$, where W_e = weight-factor). This weight-factor is calculated per km-square on the basis of the potential surface area (estimated by Klijn *et al.*, 1996; see Section 2.3.4) of the site type concerned. (No map).

Table 5.4: Spearman's rank correlation coefficients with ties - r_S - between different conservation valuation methods of km-squares, based on FLORBASE-2c data from the province of Utrecht. Methods: i - species richness; ii - sum of the species' conservation values on a 1 : 31-scale; iii - the same, on a 1 : 153-scale; iv - the same, on a WAFLO-scale; v - number of target species; vi - sum of the ecotope groups' completeness fractions C; vii - sum of the ecotope groups' conservation values CV; viii - DEMNAT-2.0. Highly correlating methods have been framed by a line.

Method	Method							
	i	ii	iii	iv	v	vi	vii	viii
i	1.00	0.97	0.90	0.86	0.51	0.71	0.69	0.60
ii	0.97	1.00	0.98	0.95	0.58	0.71	0.72	0.62
iii	0.90	0.98	1.00	0.99	0.62	0.67	0.70	0.61
iv	0.86	0.95	0.99	1.00	0.60	0.63	0.66	0.57
v	0.51	0.58	0.62	0.60	1.00	0.39	0.51	0.47
vi	0.71	0.71	0.67	0.63	0.39	1.00	0.96	0.90
vii	0.69	0.72	0.70	0.66	0.51	0.96	1.00	0.93
viii	0.60	0.62	0.61	0.57	0.47	0.90	0.93	1.00

5.3.2 Review of results

The rank-correlation coefficients listed in Table 5.4 are all *positive* and *very significant* ($P < 0.001$; Student's t-test on Spearman rank correlations; Sachs, 1982, p. 400). Three groups of methods may roughly be distinguished in the table. The first group is based on species richness and on species national rarity (method i-iv). With these methods high values are found in areas that are rich in gradients or heterogenous, like for example on the edge of the ice-pushed ridge the Utrechtse Heuvelrug and in the river-area (Figs. 5.7 and 5.8). It goes without saying that the more the conservation value-scale stretches, the less the results are determined by species richness: method i (1 : 31-scale) most resembles method ii, followed by method iii (1 : 153-scale) and iv (1 : 712-scale of WAFLO). However, there is a high correlation between the methods and from this it may be concluded that the result is *not very susceptible to the length of the conservation value-scale* and is *mainly determined by the species richness* of the km-squares. The incorporation of conservation values of species adds relatively little to the final result.

The second 'group' only contains method v, which has the number of target species per km-square as valuation measure. Especially the Utrechtse Heuvelrug itself scores high this time (Fig. 5.9). There is little correlation with all other methods.

The third group is based on the ecotope groups of Chapter 4: method vi-viii. Highly valued areas are predominantly found in fen-areas such as the Noorderpark, but also the areas along the river Vecht and along the western part of the river Rijn stand out (Fig. 5.10). There is a high correlation among the three methods.

5.4 Discussion

5.4.1 Rarity as a criterion for conservation value

Thus far, I deliberately did not consider the objectives of nature conservation policy. I only dealt with the criteria that serve to determine the conservation value of nature in a quantitative sense. In Section 5.2.1 I argued that rarity is the major criterion for the valuation of species, vegetation types and ecosystem types. A statement that may be regarded as fairly independent of policy objectives, as the following example will illustrate. Nowadays, many Dutch biologists ascribe a high value to the criterion 'naturalness'. The revival of this criterion has particularly been inspired by the so-called 'Plan Ooievaar' ('Stork Plan'; De Bruin *et al.*, 1987), which aims at the development of nature in the forelands of the main Dutch rivers, *as spontaneously as possible.* However, when a close look is taken at what exactly is found so special in that wild nature, it are mainly the rare species. The authors of this plan claim that rare species will return, like *Cucubalus baccifer* (UK: Berry Catchfly; NL: Besanjelier), *Populus nigra* (UK: Black-poplar; NL: Zwarte populier) and even the Black stork (NL: Zwarte ooievaar), which is so rare that it does not exist in the Netherlands.

One of the reasons for appreciating rare species is that they are (potentially) endangered. Equally, an extra value would have to be attributed to species that show a negative *trend*. This could be done by predicting the future occurrence of species on the basis of trend observations (compare Gremmen, 1986), and then submit the resulting rate to Eq. (5.1). Unfortunately, as no exact data are available, for the time being we have no choice but to exclude the criterion 'trend' from a quantitative valuation. In some cases UFK-figures for different periods, e.g. for 1940 and for 1990 (Van der Meijden *et al.*, 1991), have been applied to compute a trend. However, these figures have been determined with the help of flora databases that - however useful they may be for the registration of major changes - clearly suffer from inventory effects (Section 4.3.2), making them unsuitable for the accurate calculation of a trend. Besides, a fall by 1 UFK may be related to a minor change in the distribution area of the species in question, pushing it just over a borderline into a lower UFK. Likewise, a period of 50 years on end is in fact too long for drawing conclusions about any current trend. Anyone can see that our bogs have deteriorated considerably, both in area and in the number of characteristic species. But who can tell how the situation has developed in - say - the past ten years? Has the decline continued, is there a status quo, or have the applied policy-measures perhaps resulted in a slight progress?

Trend may also be estimated on the basis of existing knowledge about the functioning of ecosystems. It may for instance be assumed that dry and nutrient-poor ecosystems are threatened by the high atmospheric deposition of nitrogen. But these are but rough estimations and it is doubtful whether there is enough scientific support in any case. According to Marguless & Usher (1981) it is common practice to give species the predicate 'endangered' for no other reason than their rarity. Besides, very rare species may show a negative tendency because the sources of immigration and recolonization have partly or completely dried up. In that case there is a correlation between rarity and trend, so that one factor in a valuation procedure is counted double.

5.4.2 Valuation of kilometer squares

In the second part of this chapter (Section 5.3) I used a number of valuation methods for the drawing of maps that show the botanical value of the province of Utrecht. Some of these methods appear to differ to a considerable extent. According to one method the edges of the Utrechtse Heuvelrug are highly valued, whereas another method emphasizes the Utrechtse Heuvelrug itself, or the fen-areas. To a number of experts in the botanical field (C.L.G. Groen of the FLORON foundation; R. van der Meijden of the National State Herbarium; E.J. Weeda, J.H.J. Schaminée and G. van Wirdum of the Institute of Forestry and Nature Research IBN-DLO). I made the request to pass a judgement by presenting them with the maps[1] and the following question: "which map is the best, according to your personal judgement, and which one the worst; could you indicate an order of merit?". The study was carried out single blind and the experts were consulted independently of one another.

They unanimously gave the lowest qualification to the map that is based on target species (Fig. 5.9, method v). This poor result is caused by the procedure that was used to derive the target species. The fact that all species that have gone back with only 1 UFK between 1940 and 1990 are already labeled as 't-species' may illustrate this. Another example of the inadequate selection procedure is the selection of i-species: this selection is carried out in such a peculiar way (Van Beers, 1993) that species like *Quercus robur* (UK: Pedunculate oak; NL: Zomereik) and *Cynosurus cristatus* (UK: Crested Dog's-tail; NL: Kamgras) - which are by no means rare in international respect - are already included in that category. International rarities like *Isoetes lacustris* and *I. echinospora* (UK: Quillwort and Spring Quillwort; NL: Grote en Kleine biesvaren), on the other hand, are not.

The three conservation value maps based on the ecotope maps (method vi-viii) were qualified as the best of the set, out of which method vii was preferred by four out of the five experts. Method vii comprises practically all criteria that according to Margules & Usher (1981) are common in conservation value studies and can also fairly reliably be determined: diversity, rarity, naturalness and area. *Diversity* is manifest both in the relative diversity per ecotope group (completeness) and in the diversity of ecotope groups within a km-square. *Rarity* is manifest in a conservation value per ecotope group that is based on rarity. According to Margules & Usher (1981), *naturalness* is more difficult to determine; they claim that the best way to do this is probably by considering the contribution of indigenous wild plant species. Naturalness has been incorporated in method vii by

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[1] In all fairness I have to add that the maps presented to these experts were slightly different from the maps discussed in this publication, as they were based on FLORBASE version 1 in stead of version 2c.

calculating exclusively with the data of wild plants and also by omitting several ecotope groups of moist/dry sites that are either moderately rich or very rich in nutrients. Dry sites are usually nutrient-poor by nature; the distribution maps in question mainly show the influence of eutrophication. Finally, the criterion *area* has not been incorporated in method vii. This does not really matter, since inventory entities of a fixed size of 1 km^2 have been used. Adriani & Van der Maarel (1968) and Dony & Denholm (1985) give a method that allow inventory entities that differ in size to be compared.

So method vii was favored and, consequently, it is this method that has taken the place of method viii in the version 2.1 of the DEMNAT model.

5.4.3 Valuing vegetation relevés

The results of this research are based on data per km-square but whether or not they are also relevant for even more detailed scale levels remains to be seen. To find out about this, I examined the methods i-vii with the help of a database of 1695 vegetation relevés from the province of Drente (Dijkstra *et al.*, 1992) that I could coincidentally get hold of. Because of the fact that the threshold values T in the ecotope maps have been determined for km-squares instead of vegetation relevés, I lowered the first threshold value T_1 for this research to $0.1T_1$ and the third one to $0.9T_3$. The rank correlation matrix, which was then derived from the relevé database, appears to coincide roughly with Table 5.4. In other words: *also on the level of relevés the methods i-iv are alike, as well as the methods vi and vii, whereas method v resembles none of the other ones.*

Additionally, a ninth method (method ix) was dealt with in this brief research. This method was developed specially for vegetation relevés by Hertog & Rijken (1992), see also Heijmans (1996), Schouwenberg *et al.* (1997) and Van der Sluis (1996). According to Hertog & Rijken their method produces results that are generally acceptable to an expert. This statement was confirmed in a small study where 10 relevés were judged both by experts and by quantitative valuation methods (Van der Sluis, 1996). In method ix the value of a relevé is determined by: (1) the conservation value of the species (based on national and international rarity, trend, vulnerability, indignity, representativeness), (2) the species cover, and (3) the species richness. As was the case with the methods vi-vii, method ix takes into account that there are certain vegetation types that have little species richness by nature.

Of all methods, method vii appeared to correlate most with method ix ($r_S = 0.82$), in spite of the fact that for method vii a conservation value scale was used (Table 5.3) that is actually too short for vegetation relevés, and in spite of the fact that the threshold values for all ecotope groups were simply lowered with the same factor. Each of the species-based methods i-v showed far less correlation with method ix. The rank-correlation coefficient r_S with the WAFLO-method was for instance as low as 0.44.

Hence, also on the level of relevés *an approach that allows for the relative diversity* *of the ecosystem type or vegetation type, produces the most satisfactory results.* At least to experts. But it remains a matter of taste: everyone is free to have a greater appreciation for the coniferous 'Utrechtse Heuvelrug' than for the 'Noorderpark' with its fen-vegetation.

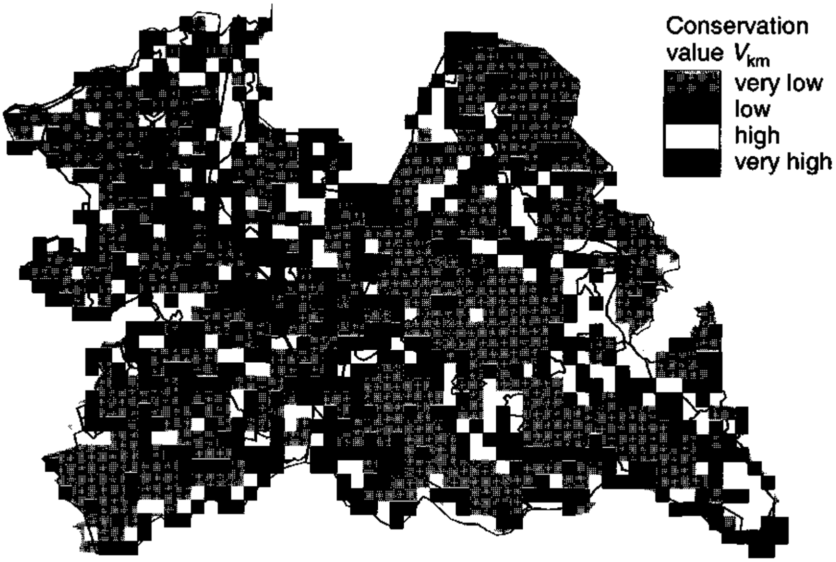

Figure 5.7: Botanical conservation values in the province of Utrecht according to method i: the number of species (species richness) per km-square. Based on FLORBASE-2c (without gap-filling).

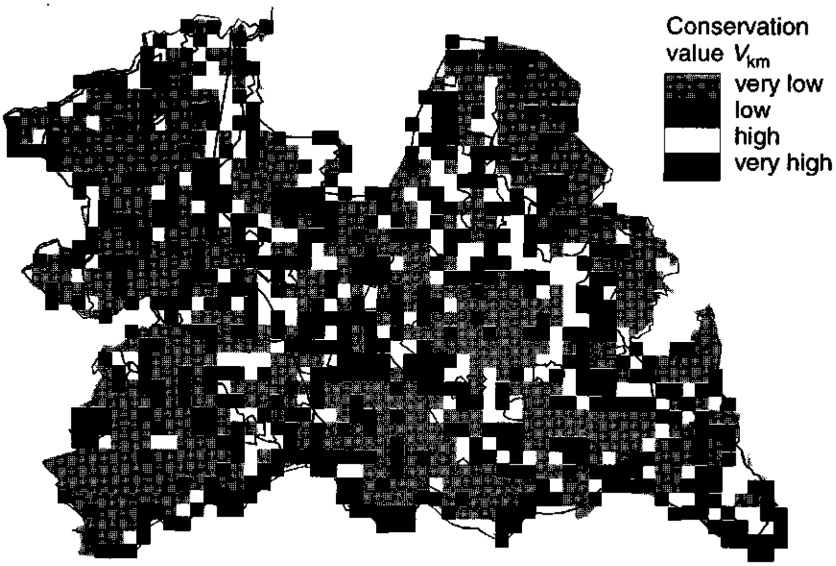

Figure 5.8: Botanical conservation values in the province of Utrecht according to method iv (WAFLO method): the sum of species conservation values per km-square. Based on FLORBASE-2c (without gap-filling).

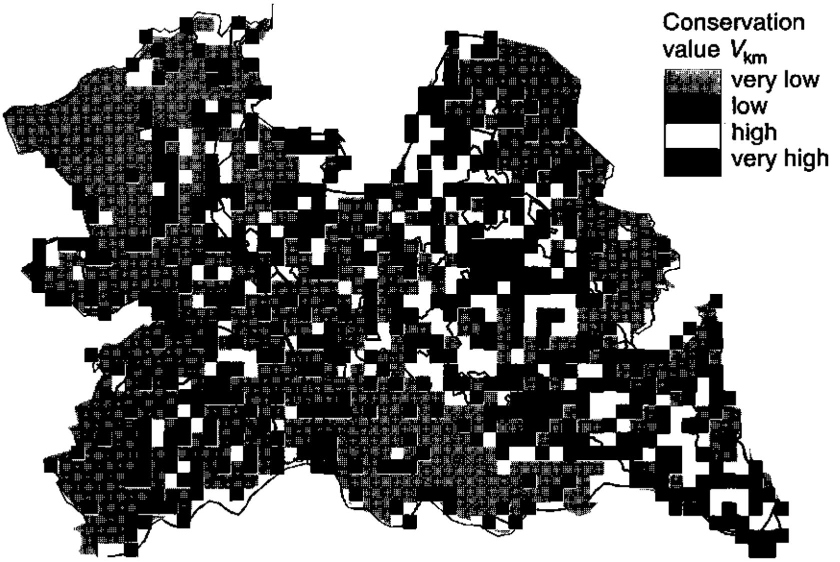

Figure 5.9: Botanical conservation values in the province of Utrecht according to method v: the number of target species per km-cell. Based on FLORBASE-2c (without gap-filling).

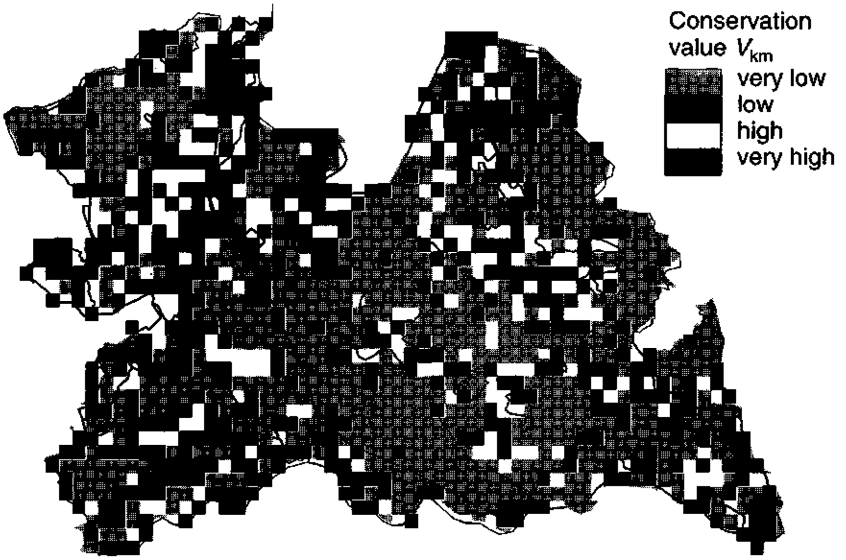

Figure 5.10: Botanical conservation values in the province of Utrecht according to method vii: sum of ecosystem conservation values per km-square. Based on FLORBASE-2c (without gap-filling).

Summary and conclusions

Starting point: the 'desiccation' of the Netherlands

In the Netherlands, nature is largely adapted to wet circumstances as may be illustrated by typical landscapes, such as brook-valleys, swamp woodlands, fens, bogs, dune slacks, wet heathlands and salt-marshes. This wet character has everything to do with the low surface of the country, which causes the groundwater level to be rather close to the roots of the plants. However, especially during the second half of this century, groundwater levels fell, mainly as a result of intensified drainage of agricultural land and a steadily increasing extraction of groundwater. To prevent the levels from becoming unacceptably low, Rhine and Meuse water has been let into the land. This river water is often 'area-alien', which means that its chemical composition differs from the original waters in the area.

As a consequence of the groundwater fall and the inlet of area-alien water, nature deteriorated. The past few decades, this *desiccation* has been mentioned both by researchers and groundkeepers as one of the main causes for the deterioration of nature. The Government decided to do something about it: in 1990 a resolution was adopted, which states that, by the year of 2000, the desiccated area has to be diminished with at least 25%. Policy measures against desiccation were laid down by the Dutch Government in several policy documents, such as the National Policy Plan on Water Management and the National Policy plan on Drinking Water and Industrial Water Supply. These documents are based on extensive studies, some of which form the starting point of this publication.

FLORBASE and the ecotope system

When predictions are needed concerning the effects of certain proposed measures on nature, one has to be provided with information about various types of nature in various parts of the Netherlands. In this publication, such information was derived from the database *FLORBASE*, which contains the observation records over the period 1975 – 1995 concerning all vascular plants that grow in the wild. In FLORBASE, the data are stored in a grid, consisting of 1 × 1 km cells (*kilometer squares*). FLORBASE was used for the making of 28 *ecosystem maps*. These are distribution maps showing the occurrence and the botanical quality (*completeness*) of ecosystem types per kilometer square. The ecosystem maps served as geographical input to the ecohydrological model *DEMNAT* (Dose Effect Model for terrestrial NATure).

In accordance with for instance objectives and map scale, there are various methods for the division of nature in ecosystem types. In this publication the *ecotope system* of the Leiden University was used. The ecotope system contains a classification of ecosystem types on the basis of five characteristics that explain important differences in the species composition of the plant cover of the Netherlands: Salinity, Moisture regime, Nutrient availability, Acidity (all four abiotic) and Vegetation structure. For each of these characteristics, a number of classes have been distinguished. The characteristic Moisture regime, for example, is divided into the classes Aquatic, Wet, Moist and Dry. Combinations of classes result in so-called *ecotope types*: these are the ecosystem types of the ecotope system. One ecotope type is e.g. a Grassland on a Dry, Nutrient-poor and Alkaline soil. Another, a Tall herb vegetation on a Brackish, Wet and Very nutrient-rich soil. By combining those classes that are abiotic, the *site types* of the ecotope system are obtained (example: Brackish, Wet, Very nutrient-rich).

The ecotope system also contains an allotment of all plant species of the Netherlands to ecological species groups. Each of these groups is composed of the plant species that are indicative for the corresponding ecotope type. Species may be allotted to several ecological groups, allowing the system to account for the indicative value of species. Hence, species that are choosy as to their site are ascribed to one single ecotope type, whereas the 'easy' species are ascribed to many.

The reason that, in this publication, the ecotope system was used, is that - compared to other classification systems - it has a major advantage: the types have been defined according to abiotic factors that may be directly affected by environmental changes. As a result, the effects of these changes (e.g. eutrophication, desiccation and acidification) may relatively easily be translated into changes in the vegetation's species composition. For example, a fall in the groundwater level often results in a lower availability of moisture to the plants, an accelerated mineralization of organic matter (which releases nutrients) and a reduced influence of alkaline groundwater in the root zone. This comes down to changes in respectively the moisture regime, the nutrient availability and the acidity of the site. These are precisely the three classification characteristics of the ecotope system.

The influence of hydrological measures on nature (Chapter 2)

In order to predict the effect of water management on nature, a number of ecohydrological models is presently available in the Netherlands. For practical reasons, these models are strong simplifications of reality. For instance, only the way the *vegetation* reacts on hydrological changes is calculated, whereas no attention is paid to the implications for animal life. This is generally deemed justifiable, since plants react on hydrological changes in a more direct way. Besides, it is normally assumed that the vegetation is in balance with its site, and that this site is homogeneous, without e.g. any vertical stratification in acidity and nutrient richness. In this publication, a rather negative judgement is given on totally statistical models. Practically all species on earth are rare from a statistical point of view, causing them to be easily overlooked in a sample collection according to chance. Nevertheless, such random sampling is necessary for a statistical approach. Preference is given to a model approach, in which exclusively those factors and those processes are incorporated, of which the importance for the vegetation is taken for a fact beforehand.

The ecohydrological model DEMNAT is able to predict changes in the completeness of ecosystem types as a result from changes in water management in the Netherlands: the model may be said to compute new ecosystem maps. To facilitate the judgement on scenarios, the changes are also expressed in conservation value figures. Conservation valuation clears the way for the summing up of outcomes of separate ecosystem types, upon which they may be reflected by one single map or by one single conservation value figure for all of the Netherlands.

The makers of DEMNAT tried to construct a model that is very practical in use. Unlike some other hydrological models, DEMNAT permits the calculation of the hydrological input by *existing* hydrological models. The consequences of hydrological measures - e.g. the reduction of groundwater extraction or the rising of surface water levels - are translated by hydrological models in terms of changes in: (1) the spring groundwater level, (2) spring water levels of small surface waters, (3) the intensity of upward seepage. These three variables form the hydrological input for DEMNAT.

DEMNAT itself consists of three parts: (i) a *geographical schematization* of the Netherlands, (ii) *dose-effect relations*, and (iii) a *conservation valuation module.*

Various maps were incorporated in the geographical schematization (i). One map, for example, incorporates the calculation units of the hydrological models that feed DEMNAT. Furthermore, there are of course the ecosystem maps, with the resolution of 1 km^2. And another map to be mentioned is the $1:50,000$ soil and groundwater depth map of the Netherlands, which DEMNAT uses for further localization of ecosystem types within the kilometer squares. However, this map is especially important because the effects of hydrological changes on the vegetation is largely dependent on the type of soil. For example: in a fen soil, a lowering of the groundwater level leads to a stronger mineralization of organic matter than in a humus-poor sand soil. An overlay of all maps leads to the basic calculation units of DEMNAT: the *ecoplots* (size: $0.25 - 100$ ha).

With dose-effect relations (ii) it is calculated for every ecoplot how the completeness of an ecosystem type changes (the effect) as a result of the three hydrological input variables (the doses). These relations were determined for every possible combination of ecosystem type and soil type. For this, changes in moisture regime, nutrient richness and acidity were assessed for every soil type and, subsequently, the impact of these changes was translated into changes in the completeness of the ecosystem type.

Finally, the completeness changes may, on the basis of the conservation valuation module (iii), be expressed in changes in conservation value figures.

The main merits of DEMNAT are that (1) the model contains every element that is necessary for an adequate prediction, (2) that the geographical data are carefully attuned to one another and (3) that the model smoothly connects to hydrological models. This does not mean that reliable results can be guaranteed. DEMNAT is likely to be a good predictor of *damage* to nature. For this purpose, it can also be used for *parts* of the Netherlands, such as the provinces. However, just like all other ecohydrological prediction models, it is less suitable for the prediction of the *recovery* of nature. For better calculations of such, an improvement of the geographical schematization should be the first step. For this, data about the height of the soil surface, about the vegetation structure and about the level of atmospheric nitrogen deposition, might be of use.

The descriptive capacity of the ecotope system (Chapter 3)

On the ecosystem maps of DEMNAT (described in Chapter 4), not only the occurrence of ecosystem types has been indicated for every kilometer square, but also their completeness. Therefore, all of them together describe the spatial variation in the plant cover of the Netherlands.

However, originally, the ecotope system was designed for applications in nature-effect studies, in stead of for the description of the vegetation in a certain area. For this, Dutch vegetation scientists often use a division, which was obtained by the method of *phytosociology* ('French- Swiss School'). In the phytosociology, *vegetation relevés* (sample stands from the plant cover, varying from a few to several dozens of square meters) are used for the derivation of the basic vegetation units: the *associations*. The phytosociological classification system is hierarchical, meaning that associations are combined into *alliances*, which, in their turn, form *orders*, and so on. An overview of all phytosociological units (types) in the Netherlands, with their associated species, was published in 1969 by Westhoff & Den Held. The past few decades, this publication served as an important work of reference for phytosociological research in the Netherlands.

Chapter 3 deals with the question whether or not the ecotope system is suitable for a FLORBASE-aided description of the vegetation of the Netherlands. To this end, the division in ecological groups was compared to the allotment of plant species to phytosociological vegetation-units according to Westhoff & Den Held. The comparison is based on two conditions that have to be met by a reliable classification system and its species division. The first condition is that, averagely spoken, species within one and the same classification unit should show more resemblance as to their distribution pattern, than is the case when they are compared to species from different units. In other words, the units should have a relatively high *internal correlation.* The second condition is that - as to their distribution - species of one unit should be clearly distinguishable from species of all other units. A low *external correlation*, i.e. a high *distinguishing capacity*, is necessary for the description of the total variation in the plant cover of the Netherlands.

In the comparison, the internal and the external correlations of the classification units were calculated from FLORBASE-data. The outcome of this calculation is that - as to internal correlation - the ecotope system is comparable to the phytosociological system (both on association level and on the level of alliances). The distinguishing capacity of the ecotope system, on the other hand, appears to be significantly higher than that of the phytosociological system. The conclusion to be drawn is that the ecotope system is suitable for the FLORBASE-aided description of the plant cover of the Netherlands. And not only that. Surprisingly, it also appears to be of better use for this than the phytosociological division by Westhoff & Den Held.

Besides, it follows from the analysis that the ecotope system is consistent: two ecotope types, having a high external correlation, often belong to one and the same site type, which means that they only differ in vegetation structure. On the other hand, the results hardly reflect the hierarchy of the phytosociological division: highly correlated associations seldom originate from the same alliance; highly correlated alliances in their turn, seldom originate from the same order.

Distribution maps of ecosystem types (Chapter 4)

In this publication, 28 maps are presented, showing the completeness per kilometer square of a corresponding number of ecosystem types (Figs. 4.26-4.53). These ecosystem types are based on the ecotope system. For practical reasons, the ecotope types that are distinguished by the ecotope system have been aggregated into so-called *ecotope groups*, i.e. combinations of ecotope types that differ in vegetation structure, whereas they belong to the same site type. The maps were made in three steps.

The first step served the allotment of species to ecotope groups by means of *indicator values*. The higher the indicator value of a particular species, the more indicative that species is for the ecotope group in question.

The inventory of the Netherlands has not been carried out with the same intensity everywhere, causing the distribution maps that are directly based on FLORBASE to show spatial distortions. Therefore, the second step was needed, serving the completion of species by means of *gap-filling* in every ecotope group. This method is based on the phenomenon that some species may function as *guiding species* for other species. If a guiding species is found somewhere, the presence of its accompanying species may be taken for granted. Gap-filling led to a second flora database, which is more complete and, as a result, yields ecosystem maps that are more reliable.

In the third step, ecosystem maps were derived from the gap-filled database. For this, the indicator values per kilometer square were added up for every ecotope group, resulting in a *score* that provided information about the presence and the botanical quality of the ecotope group in question. However, scores of different ecotope groups cannot be compared directly. One reason for this is that every ecotope group has its own particular number of indicator species. To make up for this, *threshold values* were introduced, dividing the scores into four completeness classes. These threshold values have been determined for every single ecotope group. Scores that are lower than the first threshold value are disposed of as 'noise'. Only when the score surpasses this threshold value, may the ecotope group supposed to be present and may it be used as a measure for botanical quality. The second, third and fourth completeness classes have been given the qualifications 'low', 'high' and 'very high'.

Ample attention was paid to the question whether gap-filling risks the chance of more errors being introduced than that there are corrected. To this end, various tests were applied on old flora databases (dating from the inventory periods 1902–1950 and 1950 – 1980). These tests show that gap-filling is justified and improves the ecosystem maps to a considerable extent.

For that matter, one of the tests revealed some interesting facts about inventory making in former times. In the beginning of this century, inventories were mainly made in and around one's home town. Wet and nutrient-rich places were avoided, which may be explained by the fact that - to most florists - nutrient-rich places are of little interest, since they usually only harbor common species that are not very 'critical'. There is, however, a very simple and more convincing explanation: the florists avoided wet places because they had no boots! It was not until the fifties, after all, that affordable, mass-produced rubber boots were available on the market. Furthermore, it appeared that, during the second half of this century, the quality of the inventories was to a large extent determined by the province borders. This may be explained by the fact that since the seventies provincial authorities carried out inventories, each of them using its own method.

Chapter 4 also deals with the way threshold values were established by expert judgement. This judgement appears to have been systematic to the extent that an imitation by mathematical procedures is justified, not only for the kilometer squares, but also for larger grid cells, measuring for instance 5 × 5 km. For the application of the ecosystem maps in policy analysis, e.g. carried out with DEMNAT, this is a gratifying result. An inconsistent judgement, after all, could - for instance - lead to the situation that regions with many over-estimated ecotope groups, would dominate the model results.

Finally, the ecosystem maps were compared to other data about the presence of nature in the Netherlands. Insofar as the comparison permits conclusions, it may be stated that the maps are reliable.

Nature valuation (Chapter 5)

For applications in e.g. prediction models, it may be useful to express the subjective value that people ascribe to nature, in figures. Chapter 5 deals with the quantitative valuation of nature. A distinction is made between the valuation of *classification units* - such as species, vegetation types and ecosystem types - and the valuation of *actual areas.*

In most studies the valuation of classification units is based on the criterion *rarity*: "The more rare, the more valuable" is a generally applied assumption. But apart from fundamental reasons for giving a high valuation to rare classification units (e.g.: "humanity should protect rare species because they are threatened by extinction"), there is also a practical reason for this: 'rarity' is the only valuation criterion for classification units that can be determined with reasonable reliability. The measuring of rarity is usually done by placing a grid of rectangular cells on the area in question. For the Netherlands, for instance, the rarity of plant species was determined by counting in how many 5 kilometer squares each species was found. Chapter 5 shows that the outcomes of such countings are influenced by the grid cell size. The larger the cells, the less variation in the degree of occurrence between rare and common species. Moreover, cell-enlargement may also cause shifts in the order of rarity of species.

Chapter 5 presents a mathematical formula, with which the value of classification units may be expressed in figures on a cardinal scale. This valuation formula is based on two assumptions. The first assumption, which is far from new, since it has been used in many other quantitative valuation methods, comes down to the following: the value of a classification unit increases as it becomes more rare. In simple words, the rare Bee Orchid is more valuable than the common Dandelion. The second assumption, which I introduced in this publication, is that an increase of a certain classification unit should be judged upon in a positive sense: increase of the number Bee Orchid populations should be welcomed. The latter assumption implies that, in a national accounting of conservation values, the common species would outweigh the rare species because of their numerousness, even though the value of their individuals is lower.

The valuation formula served the calculation of conservation value figures for separate species and, in addition to that, of potential conservation values for the ecotope groups of Chapter 4.

In the second part of Chapter 5 the botanical conservation value of actual areas was paid attention to. Eight valuation methods were compared to one another, on the basis of FLORBASE. The comparison focused on the province of Utrecht, since the flora inventory of this area was carried out thoroughly. For each method, a color map was made, showing the conservation value per kilometer square (Figs. 5.7-5.10).

One method is for example based on the criterion 'species richness': the more species occur in a certain kilometer square, the more valuable that square is. Other methods start by ascribing a conservation value to species, before adding them up. However, the drawback of these methods is that *all* species, including those that are part of disturbances, add to the area's conservation value in a positive sense. Another drawback is that ecosystems that are poor in species by origin - such as bogs and salt marshes - get too low a valuation. On the 28 ecosystem maps of Chapter 4 these objections have been met. These maps may be combined into one single conservation value map by calculating one value per kilometer square, which is based on: (1) the number of ecotope groups in a kilometer square, (2) the completeness of these ecotope groups, and (3) the potential conservation value of these groups.

The eight conservation value maps were submitted - 'single blind' - to the judgement of five experts, who were consulted separately of one another. All five chose the conservation value map that is based on ecosystem maps as the most adequate. It is remarkable that they unanimously rejected the method that is based on the number of *target species* per kilometer square. These target species were selected by the Ministry of Agriculture, Nature and Fishery to facilitate the evaluation of results of nature management and nature policy.

The eight valuation methods were also compared on the basis of a large database of vegetation relevés. The results appeared to correspond with the results on the level of kilometer squares: also on the level of relevés the ecotope group-based valuation method turned out to be the best.

Samenvatting en conclusies

Aanleiding: de verdroging van Nederland

De natuur in Nederland is grotendeels aangepast aan natte omstandigheden. Denk aan de Nederlandse beekdalen, moerasbossen, rietlanden, poldersloten, hooilanden, duinvalleien en kwelders. Dat natte karakter hangt natuurlijk samen met de lage ligging van het land; daardoor staat de grondwaterstand dichtbij de plantewortels. Nederland is echter - vooral in de tweede helft van deze eeuw - droger geworden. Met name ten gevolge van peilverlagingen in landbouwgebieden en de winning van grondwater. Om te lage grondwaterstanden te voorkomen wordt heden ten dage op veel plaatsen water uit de Rijn of de Maas aangevoerd. Dit rivierwater is meestal 'gebiedsvreemd': het heeft dan een andere chemische samenstelling dan het oorspronkelijke water.

Het gevolg van de grondwaterstandsdalingen en de inlaat van gebiedsvreemd water was dat de natuur verpieterde. Al tientallen jaren wordt 'verdroging' door zowel terreinbeheerders als onderzoekers genoemd als een van de belangrijkste oorzaken voor de achteruitgang van de vegetatie. De Overheid wil wat aan die verdroging doen. Zo is in 1990 in de Tweede kamer een motie aangenomen waarin staat dat het oppervlak verdroogde natuur in Nederland in het jaar 2000 met tenminste 25% moet zijn terug gedrongen. Het anti-verdrogingsbeleid wordt door de Overheid vastgelegd in diverse beleidsdocumenten, zoals de Nota Waterhuishouding en het Beleidsplan Drink- en Industriewatervoorziening. Aan de in die documenten voorgestelde maatregelen liggen vaak uitvoerige studies ten grondslag. Enkele van deze studies vormen de aanleiding voor deze publicatie.

FLORBASE en het ecotopensysteem

Wanneer men wil kunnen voorspellen wat de effecten van voorgenomen maatregelen op de natuur zullen zijn, dan zal men eerst moeten weten waar in Nederland welk type natuur voorkomt. In deze publicatie is die informatie afgeleid van het bestand *FLORBASE*, dat de vondstgegevens uit de periode 1975 – 1995 bevat van alle in het wild voorkomende vaatplanten. De gegevens in FLORBASE zijn opgeslagen in een raster met cellen van 1 × 1 km (*kilometerhokken*). In deze publicatie is FLORBASE gebruikt voor het maken van 28 *ecosysteemkaarten*: verspreidingskaarten met het voorkomen en de botanische kwaliteit (*volledigheid*) van ecosysteemtypen per kilometerhok. De ecosysteemkaarten dienen als geografische invoer voor het ecohydrologische model *DEMNAT* (Dosis-EffectModel NAtuur Terrestrisch).

Afhankelijk van onder meer doelstellingen en kaartschaal, kan men de natuur op verschillende manieren indelen in ecosysteemtypen. In deze publicatie is gebruik gemaakt van het Leidse *ecotopensysteem*. Het ecotopensysteem bevat een classificatie van ecosysteemtypen op basis van kenmerken die belangrijke verschillen in de soortensamenstelling van het plantendek van Nederland verklaren. Dit zijn de vier abiotische kenmerken Zoutgehalte, Vochttoestand, Voedselrijkdom en Zuurgraad, en het kenmerk Vegetatiestructuur. Ieder van deze kenmerken is ingedeeld in een aantal klassen. Het kenmerk Vochttoestand bijvoorbeeld, in de klassen Aquatisch, Nat, Vochtig en Droog. Combinatie van klassen resulteert in zogenaamde *ecotooptypen*, de ecosysteemtypen uit het ecotopensysteem. Een ecotooptype is bijvoorbeeld een Grasland op een Droge, Voedselarme, Basische bodem, of een Ruigte op een Brakke, Natte, Zeer voedselrijke bodem. Wanneer alleen de abiotische klassen worden gecombineerd ontstaan de *standplaatstypen* van het ecotopensysteem (voorbeeld: Brak, Nat, Zeer voedselrijk).

Het ecotopensysteem bevat ook een toedeling van alle Nederlandse plantesoorten aan *ecologische groepen*. Zo'n ecologische groep is samengesteld uit alle plantesoorten die indicatief zijn voor het ermee corresponderende ecotooptype. Soorten kunnen bij meerdere ecologische groepen zijn ingedeeld. Op die manier houdt het ecotopensysteem rekening met de indicatieve waarde van soorten. Soorten die kieskeurig zijn ten aanzien van hun standplaats, zijn dus ieder bij slechts één ecotooptype ingedeeld, terwijl de 'alleseters' bij vele typen zijn ingedeeld.

In deze publicatie is het ecotopensysteem gebruikt omdat dit ten opzichte van veel andere classificatiesystemen een belangrijk voordeel heeft, namelijk: dat de typen zijn gedefinieerd aan de hand van abiotische factoren die direct kunnen worden beïnvloed door veranderingen in het milieu. De effecten van deze veranderingen - denk aan vermesting, verdroging en verzuring - zijn daardoor relatief eenvoudig te vertalen in veranderingen in de soortensamenstelling van de vegetatie. Zo resulteert een verlaging van de grondwaterstand dikwijls in een verminderde beschikbaarheid van vocht voor de planten, een versnelde mineralisatie van organische stof waarbij voedingsstoffen vrijkomen, en een afname van de invloed van basenrijk grondwater in het wortelmilieu. Zo'n verlaging leidt derhalve tot veranderingen in respectievelijk de vochttoestand, de voedselrijkdom en de zuurgraad van de standplaats. En dit nu, zijn precies drie classificatiekenmerken van het ecotopensysteem.

Natuureffecten van hydrologische ingrepen (Hoofdstuk 2)

In Nederland bestaan er verschillende ecohydrologische voorspellingsmodellen, waarmee kan worden berekend hoe de natuur reageert op hydrologische ingrepen. Uit praktische overwegingen zijn in deze modellen vergaande vereenvoudigingen van de werkelijkheid doorgevoerd. Zo wordt alleen berekend hoe de vegetatie reageert op hydrologische veranderingen, en niet hoe de fauna dat doet. Dit acht men verantwoord omdat de vegetatie het meest direct door de waterhuishouding wordt beïnvloed. Verder wordt altijd aangenomen dat de vegetatie in evenwicht verkeert met zijn standplaats, en dat die standplaats homogeen is en bijvoorbeeld geen verticale gelaagdheid in zuurgraad en voedselrijkdom kent. In deze publicatie wordt een tamelijk negatief oordeel geveld over volledig statistische modellen. Bijna alle plantesoorten op aarde zijn in statistisch opzicht zeer zeldzaam, waardoor ze makkelijk gemist worden in een volgens het toeval uitgevoerde bemonstering. Zo'n 'random' bemonstering is echter wel vereist in een statistische aanpak. Gepleit wordt voor een modelbenadering waarin alleen díe processen en factoren zijn verwerkt, waarvan men op voorhand al weet dat ze voor de vegetatie van belang zijn.

Met het model DEMNAT kan worden berekend hoe de volledigheid van ecosysteemtypen verandert wanneer er veranderingen optreden in de waterhuishouding van Nederland: het model rekent als het ware nieuwe ecosysteemkaarten uit. Om scenario's makkelijk te kunnen beoordelen worden de veranderingen ook uitgedrukt in natuurwaardecijfers. Na natuurwaardering is het mogelijk de uitkomsten voor de verschillende ecosysteemtypen bij elkaar op te tellen, zodat ze kunnen worden weergegeven in één kaart, of in één natuurwaardecijfer voor heel Nederland.

Bij de bouw van DEMNAT is gestreefd naar een model dat in de praktijk gemakkelijk is te gebruiken. Zo kan, in tegenstelling tot sommige andere ecohydrologische modellen, de hydrologische invoer berekend worden door *bestaande* hydrologische modellen. De gevolgen van een waterhuishoudkundig scenario - bijvoorbeeld het reduceren van grondwaterwinningen, of het opzetten van slootpeilen - worden door die hydrologische modellen uitgedrukt als *veranderingen* in de voorjaarsgrondwaterstand, in het peil van kleine oppervlaktewateren, en in de kwelintensiteit. Deze drie variabelen vormen de hydrologische invoer voor DEMNAT.

DEMNAT zelf bestaat uit drie onderdelen: (i) een *geografische schematisering* van Nederland, (ii) *dosis-effectrelaties* en (iii) een *natuurwaarderingsmodule.*

In de geografische schematisering (i) zijn verschillende kaarten betrokken. In een kaart zijn bijvoorbeeld de rekeneenheden opgenomen van de hydrologische modellen die DEMNAT voeden. Verder zijn er natuurlijk de ecosysteemkaarten, met een resolutie van 1 km^2. De 1 : 50.000 bodem- en grondwatertrappenkaart van Nederland wordt door DEMNAT gebruikt om het voorkomen van ecosysteemtypen binnen de kilometerhokken nader te lokaliseren. De bodemkaart is echter vooral van belang omdat ecologische effecten van hydrologische ingrepen afhangen van het bodemtype. Zo leidt een verlaging van de grondwaterstand in een laagveengrond tot een sterkere mineralisatie van organische stof dan in bijvoorbeeld een humusarme zandgrond. Wanneer een 'overlay' van alle kaarten wordt gemaakt onstaan de basale rekeneenheden van DEMNAT, de *ecoplots* (grootte: $0,25 - 100$ ha).

Met dosis-effectrelaties (ii) wordt per ecoplot berekend hoe de volledigheid van een ecosysteemtype verandert (het effect) ten gevolge van de drie hydrologische invoervariabelen (de doses). Deze relaties zijn opgesteld voor iedere combinatie van ecosysteemtype en bodemtype. Daartoe is eerst bepaald hoe de vochttoestand, de voedselrijkdom en de zuurgraad van een bodemtype veranderen. De gevolgen van deze abiotische veranderingen zijn daarna vertaald naar veranderingen in de volledigheid van het ecosysteemtype.

De veranderingen in de volledigheid kunnen tenslotte met de natuurwaarderingsmodule (iii) worden uitgedrukt in veranderingen in natuurwaardecijfers.

De kracht van DEMNAT is dat het model alle onderdelen bevat die voor een goede voorspelling nodig zijn, dat geografische gegevens op elkaar zijn afgestemd, en, tenslotte, dat het model goed aansluit op hydrologische modellen. Dit wil niet zeggen dat het model ook betrouwbare resultaten oplevert. *Schade* aan de natuur kan waarschijnlijk goed met DEMNAT worden bepaald, ook voor *regio's* binnen Nederland (bijvoorbeeld provincies). Net als bij alle andere ecohydrologische modellen, laat de berekening van natuur*herstel* echter te wensen over. Deze berekening kan in de eerste plaats worden verbeterd door de gebiedsschematisering te verbeteren. Daarbij zou gebruik kunnen worden gemaakt van gegevens over maaiveldshoogte, vegetatiestructuur, en over het niveau van de atmosferische stikstofdepositie.

De beschrijvende kracht van het ecotopensysteem (Hoofdstuk 3)

Op de ecosysteemkaarten van DEMNAT (beschreven in Hoofdstuk 4) is voor ieder kilometerhok niet alleen het voorkomen van ecosysteemtypen aangegeven, maar ook de volledigheid van die ecosysteemtypen. De kaarten beschrijven derhalve samen de ruimtelijke variatie in het plantendek van Nederland.

Het ecotopensysteem is echter in eerste instantie ontworpen voor toepassingen in natuur-effectstudies, en niet voor het beschrijven van de vegetatie van een gebied. Daarvoor gebruikt men in Nederland vaak een indeling die volgens de methode van de *plantensociologie* ('Frans-Zwitserse school') is verkregen. In de plantensociologie worden *vegetatie-opnamen* - steekproeven in het plantendek van enkele tot tientallen vierkante meter groot - gebruikt voor het afleiden van basale vegetatie-eenheden, de *associaties*. Het plantensociologische classificatiesysteem is *hiërachisch*: associaties zijn op grond van floristische overeenkomsten samengevoegd tot *verbonden*, die op hun beurt weer tot *orden*, etc.. Een overzicht van alle plantensociologische eenheden (typen) in Nederland en de daarbij behorende soorten, werd in 1969 gepubliceerd door Westhoff & Den Held. Deze publicatie heeft decennia lang gediend als belangrijk referentiewerk in het Nederlandse plantensociologische onderzoek.

In Hoofdstuk 3 is onderzocht of het ecotopensysteem geschikt is voor de beschrijving van de vegetatie van Nederland met behulp van FLORBASE. Daartoe werd de indeling in ecologische groepen vergeleken met de indeling van plantesoorten bij plantensociologische vegetatie-eenheden volgens Westhoff & Den Held. De vergelijking is gebaseerd op twee voorwaarden die aan een goed classificatiesysteem met een bijbehorende soortenindeling mogen worden gesteld. De eerste is dat soorten van eenzelfde classificatie-eenheid - gemiddeld gesproken - qua verspreidingspatroon meer met elkaar moeten overeenkomen dan met soorten van een andere eenheid. Met andere woorden: eenheden zouden een relatief hoge *interne correlatie* moeten hebben. De tweede voorwaarde is dat soorten van eenzelfde eenheid zich qua verspreiding goed moeten kunnen onderscheiden van soorten van alle andere eenheden. Een lage *externe correlatie*, dus een groot *onderscheidend vermogen*, is nodig om de totale variatie in het plantendek van Nederland te kunnen beschrijven.

In de vergelijking werden de interne en de externe correlatie van de classificatie-eenheden berekend uit de verspreidingsgegevens van FLORBASE. Uit de berekeningen blijkt dat het ecotopensysteem qua interne correlatie vergelijkbaar is met het plantensociologische systeem (op het niveau van zowel associaties als verbonden). Het onderscheidend vermogen van het ecotopensysteem is echter significant hoger dan dat van het plantensociologische systeem. Uit deze resultaten kan worden geconcludeerd dat het ecotopensysteem geschikt is voor de beschrijving van het plantendek van Nederland met FLORBASE; verrassend is dat het hiervoor zelfs beter te gebruiken is dan de plantensociologische indeling van Westhoff & Den Held.

Bovendien volgt uit de analyse dat het ecotopensysteem consistent in elkaar zit: twee ecotooptypen met onderling een hoge externe correlatie zijn vaak van eenzelfde standplaatstype en verschillen dus alleen van elkaar in vegetatiestructuur. Daarentegen is in de resultaten weinig terug te vinden van de hiërachie in de plantensociologische indeling: hoog gecorreleerde associaties zijn zelden van hetzelfde verbond, en hoog gecorreleerde verbonden op hun beurt, zelden van dezelfde orde.

Verspreidingskaarten van ecosysteemtypen (Hoofdstuk 4)

In deze publicatie zijn 28 kaarten gepresenteerd met daarop de volledigheid per kilometerhok van evenzovele ecosysteemtypen (Fig. 4.26-4.53). Deze ecosysteemtypen zijn gebaseerd op het ecotopensysteem. De in het ecotopensysteem onderscheiden ecotooptypen zijn echter uit praktische overwegingen geaggregeerd tot zogenaamde *ecotoopgroepen*: samenvoegingen van ecotooptypen die verschillen in vegetatiestructuur maar die behoren tot hetzelfde standplaatstype. De kaarten werden in drie stappen vervaardigd.

Als eerste werden soorten met behulp van *indicatiewaarden* toegekend aan ecotoopgroepen. Hoe hoger de indicatiewaarde van een soort voor een bepaalde ecotoopgroep, des te indicatiever de soort voor die ecotoopgroep is.

Nederland is niet overal even intensief geïnventariseerd zodat verspreidingskaarten die rechtstreeks op FLORBASE worden gebaseerd, ruimtelijke vertekeningen zullen vertonen. Als tweede stap werden daarom via een speciaal ontwikkelde *hiaatopvulmethode* per ecotoopgroep soorten aangevuld op FLORBASE. Deze methode is gebaseerd op het verschijnsel dat sommige soorten *gidssoorten* zijn voor andere soorten; wanneer zo'n gidssoort aanwezig is, kan de aanwezigheid van die andere soorten gevoegelijk worden aangenomen. Hiaatopvulling leidde tot een tweede florabestand dat vollediger is en dus meer betrouwbare ecosysteemkaarten kan opleveren.

In de derde stap werden van het hiaatopgevulde bestand ecosysteemkaarten afgeleid. Daartoe werden eerst per ecotoopgroep de indicatiewaarden per kilometerhok gesommeerd. De op deze manier berekende *score* geeft informatie over de aanwezigheid en de soortenrijkdom van de betreffende ecotoopgroep. Scores van verschillende ecotoopgroepen zijn echter niet direct onderling vergelijkbaar, onder meer omdat iedere ecotoopgroep een eigen aantal indicatorsoorten heeft. Daarom werden de scores met behulp van *drempelwaarden* genormeerd tot een viertal *volledigheidsklassen*. Voor iedere ecotoopgroep werden apart drempelwaarden vastgesteld. Scores lager dan de eerste drempelwaarde zijn als 'ruis' opgevat; pas boven deze drempelwaarde is verondersteld dat de ecotoopgroep aanwezig is en is de score gebruikt als een maat voor volledigheid. De drie volledigheidsklassen boven de eerste drempelwaarde hebben de kwalificaties 'matig', 'goed' en 'zeer goed' gekregen.

Uitvoerig is stil gestaan bij de vraag of met hiaatopvulling niet meer fouten worden geïntroduceerd dan er worden opgeheven. Via verschillende toetsen op oudere florabestanden (van de inventarisatieperioden 1902 – 1950 en 1950 – 1980) is aangetoond dat hiaatopvulling verantwoord is en leidt tot een aanzienlijke verbetering van de ecosysteemkaarten.

Een van de toetsen geeft overigens aardig inzicht in de manier waarop men vroeger de flora inventariseerde. In het begin van deze eeuw werd vooral geïnventariseerd in en rond de eigen woonplaats. Natte en voedselrijke milieus werden gemeden, onder meer omdat voedselrijke milieus voor de meeste floristen niet zo interessant zijn. Daar groeien immers algemene en weinig 'kritische' soorten. Belangrijker is echter dat men natte plekken meed, omdat men niet kon beschikken over goedkope laarzen. Betaalbare, seriematig geproduceerde kunststoflaarzen, zijn namelijk pas in de jaren vijftig op de markt gekomen. Oude excursiefoto's tonen dan ook deftige mannen - soms met hoge hoed - die leren schoenen dragen en stoffen wikkels om de kuiten hebben tegen prikkende planten. Ten ene male ongeschikt voor het betreden van moerassige terreinen, dat is wel duidelijk. In de tweede helft van deze eeuw blijkt de kwaliteit van de flora-inventarisaties sterk samen te hangen met de provinciegrenzen. Een gevolg van het feit dat sommige provinciale overheden zich vanaf de jaren zeventig met het inventarisatiewerk gingen bezighouden.

Hoofdstuk 4 gaat ook in op de manier waarop de drempelwaarden met deskundigenoordeel zijn vastgesteld. Dit oordeel blijkt zeer systematisch te zijn geweest. Zodanig zelfs, dat het met een wiskundige procedure kan worden nagebootst, niet alleen voor kilometerhokken, maar ook voor grotere rastercellen, bijvoorbeeld van 5 × 5 km. Voor toepassing in beleidsanalyses, zoals met DEMNAT, is dit resultaat verheugend. Een inconsistent oordeel kan er immers toe leiden dat regio's waar overschatte ecotoopgroepen veel voorkomen, zullen gaan domineren in de modelresultaten.

De ecosysteemkaarten werden tenslotte vergeleken met andere gegevens over de aanwezigheid van de natuur in Nederland. Voor zover de vergelijking dit toelaat kan worden geconcludeerd dat de kaarten betrouwbaar zijn.

Natuurwaardering (Hoofdstuk 5)

Voor verschillende toepassingen, zoals in voorspellingsmodellen, kan het handig zijn de subjectieve waarde die mensen toekennen aan natuur in cijfers uit te drukken. Hoofdstuk 5 gaat over het op een kwantitatieve manier waarderen van natuur. Onderscheid wordt gemaakt in de waardering van *classificatie-eenheden* - zoals soorten, vegetatietypen en ecosysteemtypen - en de waardering van *concrete gebieden*.

In de meeste studies is de waardering van classificatie-eenheden gebaseerd op het criterium *zeldzaamheid*: hoe zeldzamer een soort is, des te waardevoller men hem meestal vindt. Er zijn niet alleen fundamentele redenen om zeldzame classificatie-eenheden hoog aan te slaan (bijvoorbeeld: "de mensheid moet zeldzame soorten beschermen omdat die als eerste met uitsterven worden bedreigd"), maar er is ook een practische reden voor: 'zeldzaamheid' is het enige belangrijke waarderingscriterium voor classificatie-eenheden dat redelijk betrouwbaar kan worden vastgesteld. Het meten van zeldzaamheid gebeurt doorgaans met een raster van rechthoekige cellen dat over een aandachtsgebied is heen gelegd. Zo is voor Nederland de zeldzaamheid van plantesoorten bepaald door per soort te turven in hoeveel cellen van 5 × 5 km deze is aangetroffen. Hoofdstuk 5 laat zien dat de meetuitkomsten worden beïnvloed door de celgrootte. De verhouding in de mate van voorkomen tussen zeldzame en algemene soorten neemt af naarmate men grotere rastercellen gebruikt. Maar wat bij vergroting van de cellen ook kan gebeuren, is dat de rangorde in de zeldzaamheid van soorten verandert.

Hoofdstuk 5 geeft een formule waarmee de waarden van classificatie-eenheden in cijfers op een cardinale schaal kunnen worden uitgedrukt. De *waarderingsformule* is gebaseerd op twee veronderstellingen. De eerste ligt nogal voor de hand en deze wordt dan ook in andere kwantitatieve waarderingsmethoden toegepast, namelijk dat de waarde van een classificatie-eenheid toeneemt naarmate die eenheid zeldzamer wordt. Simpel gesteld: de zeldzame Bijenorchis is meer waard dan de algemene Paardenbloem. Een tweede voorwaarde - die in deze publicatie wordt geïntroduceerd - is dat de toename van classificatie-eenheden positief moet worden beoordeeld: wanneer de Bijenorchis zich uitbreidt in Nederland dienen wij dat toe te juichen. De consequentie van de laatste veronderstelling is dat algemene soorten (classificatie-eenheden) een hogere waarde vertegenwoordigen in een nationale boekhouding van natuurwaarden dan zeldzame soorten. Individuen van algemene soorten zijn welliswaar weinig waard, maar door hun talrijkheid winnen ze het in zo'n boekhouding toch van de zeldzame soorten.

Met de waarderingsformule werden natuurwaardecijfers berekend voor soorten, en ook *potentiële natuurwaarden* voor de ecotoopgroepen van Hoofdstuk 4.

Het tweede gedeelte van Hoofdstuk 5 gaat in op de botanische waardering van concrete gebieden. Met behulp van FLORBASE werd een achttal waarderingsmethoden met elkaar vergeleken. Dat gebeurde onder meer door van iedere methode een kaart te maken met de botanische waarde van de goed geïnventariseerde provincie Utrecht (Fig. 5.7-5.10).

Eén methode is bijvoorbeeld gebaseerd op het het criterium 'soortenrijkdom': hoe meer soorten in een kilometerhok voorkomen, des te waardevoller dat hok is. Enkele andere methoden kennen eerst een natuurwaarde aan soorten toe voordat deze worden opgeteld. Al deze op het voorkomen van soorten gebaseerde methoden hebben als nadeel dat iedere soort in postieve zin bijdraagt aan de gebiedsnatuurwaarde, dus ook soorten die kenmerkend zijn voor verstoringen. Een ander nadeel is dat ecosystemen die van nature soortenarm zijn - zoals hoogvenen - een te lage waardering krijgen. Op de 28 ecosysteemkaarten van Hoofdstuk 4 zijn deze bezwaren ondervangen. Deze kaarten kunnen gecombineerd worden tot een natuurwaardekaart door per kilometerhok één waarde te berekenen die is gebaseerd op: (1) het aantal ecotoopgroepen in het kilometerhok, (2) de volledigheid van die ecotoopgroepen, en (3) de potentiële natuurwaarde van die ecotoopgroepen.

De acht natuurwaardekaarten werden 'blind' voorgelegd aan vijf deskundigen die onafhankelijk van elkaar werden geraadpleegd. Deze deskundigen wezen de op de ecosysteemkaarten gebaseerde natuurwaardekaart als beste aan. Opmerkelijk is de unanieme afwijzing van een methode, gebaseerd op het aantal *doelsoorten* per kilometerhok. Doelsoorten zijn door het Ministerie van Landbouw, Natuurbeheer en Visserij geselecteerd om de resultaten van het natuurbeheer en -beleid te kunnen evalueren.

De acht waarderingsmethoden werden ook vergeleken met behulp van een groot bestand met vegetatie-opnamen. De resultaten bleken overeen te komen met de resultaten op kilometerhokniveau en ook op opnameniveau kwam de op ecotoopgroepen gebaseerde waarderingsmethode als beste uit de bus.

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Appendix A

Bronhouders FLORBASE

De volgende standaardtekst dient elke gebruiker van FLORBASE bij publicaties te vermelden:
"FLORBASE is een bestand met plantensoort-waarnemingen op 1 × 1 kilometerhokniveau. Het bestand bestaat uit gegevens van provincies, particulieren, terreinbeherende organisaties en instituten."
Het inventariseren van de Nederlandse flora is zeer tijdsrovend werk. FLORBASE is tot stand gekomen dankzij de inspanningen van een groot aantal bronhouders. De volgende bronhouders hebben de grootste bijdrage geleverd aan versie 2c:

Provincie Groningen
Provincie Drenthe
Provincie Overijssel
Provincie Gelderland
Provincie Utrecht
Provincie Noord-Holland
Provincie Zuid-Holland
Provincie Zeeland
Provincie Noord-Brabant
Provincie Limburg
Onderzoeksinstituut Rijksherbarium / Hortus Botanicus van de Rijksuniversiteit Leiden
Stichting Floristisch Onderzoek Nederland (FLORON)
Staatsbosbeheer
Vereniging Natuurmonumenten
Werkgroep Florakartering Drenthe
Floristische Werkgroep Twente
Floristische Werkgroep KNNV Eindhoven
Natuurhistorisch Genootschap Limburg, Plantenstudiegroep Limburg
N.V. PWN Waterleidingbedrijf NH
IBN-DLO
RIZA
NJN Nijmegen

Appendix B

Classification classes of the ecotope system

Definitions and descriptions

The following definitions of classification characteristics and description of classes originates, with slight modifications, from Runhaar *et al.* (1987) and Runhaar & Udo de Haes (1994). On the basis of field surveys (Runhaar, 1989a; Runhaar, 1989b; Runhaar *et al.*, 1997b; Runhaar *et al.*, 1997c; Runhaar *et al.*, 1997d) literature and expert judgement, the classes are described in physical terms.

SALINITY (prefix)
Three classes are distinguished, based on the chloride concentration of the water:

-	*Fresh*			$[Cl^-]$	$\leq$	$1{,}000\ mg\ l^{-1}$
b	*Brackish*	1,000	$<$	$[Cl^-]$	$<$	$10{,}000\ mg\ l^{-1}$
s	*Saline*			$[Cl^-]$	$\geq$	$10{,}000\ mg\ l^{-1}$

VEGETATION STRUCTURE (capital)
Vegetation structure differs from other factors, not so much because of the fact that it is an operational site factor itself, but rather because it is the result of the operational site factors 'vegetation management' and 'time'. Furthermore, vegetation structure determines the physiological factor 'light availability', which is especially important for understory plants. Six classes have been distinguished.

W *Water vegetation*: surface water dominated by submerged and floating plants.

V *Terrestrializing vegetation*: surface water dominated by plants (helophytes) that rise above its level.

P *Pioneer vegetation*: open vegetation on initial soils or in places where the succession is hindered by physical or physiological stress.

G *Grassland s.l.*: low vegetation with herbs, bryophytes, and/or dwarf shrubs in situations where removal by mowing and/or grazing takes place.

R *Tall herbaceous vegetation*: high, dense herbaceous vegetation in places where little or no mowing or grazing takes place.

H *Woods and shrubs*: vegetation dominated by woody species that reach heights of exceeding one meter.

MOISTURE REGIME (first figure)

The term 'moisture regime' is used to characterize the oxygen-water relations in the soil. On the basis of a field survey (e.g. Figs. 1.7 and 1.8) these classes have been defined in terms of combinations of soil texture and average spring groundwater level (SGL: based on two-weeks observations in March-April 1980 – 1986) (Runhaar, 1989b).

1 *Aquatic*: permanent surface waters or surface waters that are uncovered for short periods only.

2 *Wet*: sites with permanent or periodic low oxygen contents of the soil due to a high groundwater level; $SGL \leq 20$ cm below surface.

4 *Moist*: sites with, on average, sufficient oxygen and water availability. $SGL >$ 20 cm below surface and in sandy soils less than 60-150 cm below surface (depending on the grain size and loam content).

6 *Dry*: sites with a low moisture availability in summer because of a low groundwater level and little available soil moisture.

NUTRIENT AVAILABILITY (second figure)

This characteristic refers to the availability of macronutrients (N, P, K), or rather the availability of the limiting macro-nutrient in the root zone. Three classes are distinguished: low, moderate and high. So far, the classes have not been described quantitatively, since a reliable and measurable parameter is lacking; concentrations of individual macro-nutrients obviously give little information about nutrient availability during the growing season (Boeker, 1954; Knauer, 1972; Wheeler *et al.*, 1992, *fide* Runhaar & Udo de Haes, 1994). Only for grasslands a general indication of the boundaries between the classes can be given, using parameters that are closely connected with nutrient availability: net biomass production (kg dry weight per hectare per year) and N-mineralization (kg N per hectare per year).

- *Low*: net production on average less than $4,000$ kg; N-mineralization on average less than 50 kg. This class is always combined with an acidity class (indicated with the figures $1-3$).

7 *Moderate*: net production on average between $3,000{-}8,000$ kg; N-mineralization on average between 30 and 160 kg.

8 *High*: net production on average more than $7,000$ kg; N-mineralization on average more than 160 kg.

ACIDITY (second figure)

The acidity of the soil and of the groundwater influences vegetation in various ways. In acid soils the toxicity of Al^{3+}-ions seems to play an important role in determining species composition (Clymo, 1962; Sparling, 1967, *fide* Runhaar & Udo de Haes, 1994), while on alkaline sites the availability of Fe can be limiting for many species (Grime & Hodgson, 1968, *fide* Runhaar and Udo de Haes, 1994).

Furthermore, soil acidity influences bacterial activity in the soil and the solubility of e.g. phosphate. Three classes are distinguished.

1 *Acid* $\text{pH-H}_2\text{O} \leq 4.5$
2 *Neutral (weakly acid)* $4.5 < \text{pH-H}_2\text{O} < 6.5$
3 *Alkaline* $\text{pH-H}_2\text{O} \geq 6.5$

Comparison with other indicator lists

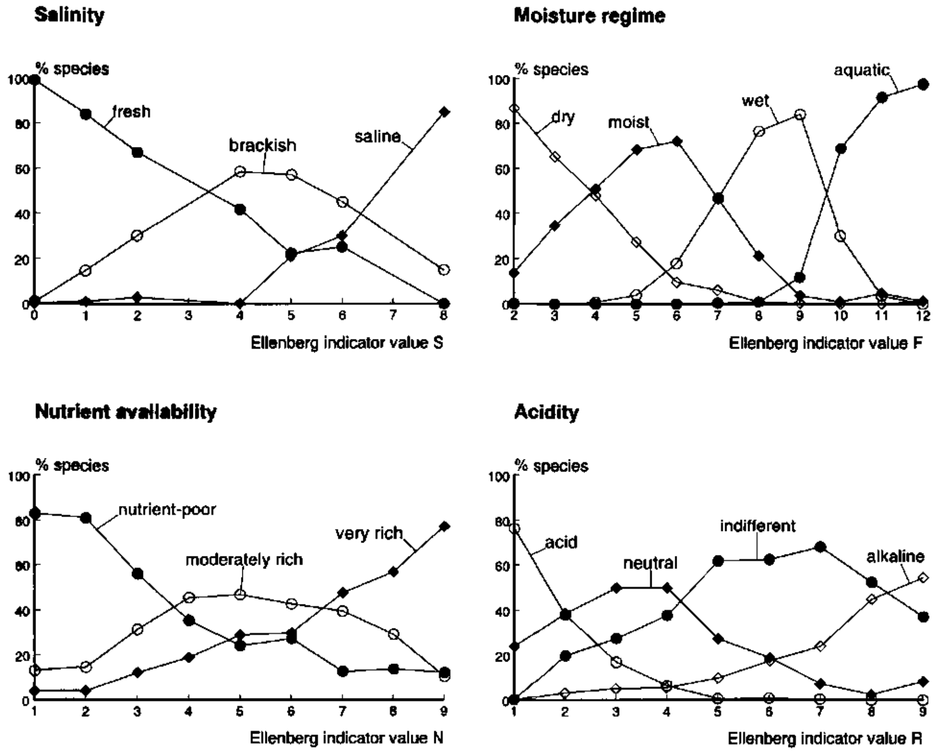
Runhaar *et al.* (1987) made a comparison of the classes of the ecotope system with the indicator value lists of Ellenberg (1979) and of Klapp (1965), see Table B.1.

The frequency-distributions of Fig. B.1 show how the classes of the ecotope system correspond with the indicator lists of Ellenberg from 1991. I constructed these frequency distributions on the basis of information of all of the species of the Dutch flora. For species ascribed to more than one class by Runhaar *et al.* (1987), I used weight factors. *Lysimachia vulgaris*, for instance, has been ascribed to 5 'wet' ecotope types and 2 'moist' ecotope types, and so this species got a weight factor of $\frac{5}{7}$ for the class 'wet', and $\frac{2}{7}$ for 'moist'.

From Fig. B.1 it appears that both indicator system correspond well with regard to the factors of 'salinity', 'moisture regime' and 'acidity'. The correspondence with regard to 'nutrient availability', on the other hand, is rather poor.

Table B.1: Relation between the classes of the ecotope system and the indicator values of moisture regime, nutrient-availability and acidity according to Ellenberg (1979) and Klapp (1965). After Runhaar *et al.* (1987).

	Moisture regime		
	Dry	Moist	Wet
Ellenberg F	1-4	4-6	7-10
Klapp F	1-3	3-6	7-9
	Nutrient availability		
	Nutrient-poor	Moderately nutrient-rich	Very nutrient-rich
Ellenberg N	1-3	4-7	7-9
Klapp N	1-3	2-4	4-5
	Acidity		
	Acid	Neutral	Alkaline
Ellenberg R	1-3	3-7	7-9
Klapp R	1-2	2-4	4-5


Salinity

Moisture regime

Nutrient availability

Acidity

Figure B.1: Frequency distributions of the classes of the ecotope system (Runhaar *et al.*, 1987) over the indicator values of Ellenberg (1991). The sum of the ecotope classes of each indicator value has been scaled to 100%. Ellenberg's Salinity indicator value 1, for example, consists of 84% fresh, 15% brackish and 1% saline species. Indicator values with less than 10 species have been omitted.

Appendix C

The French-Swiss school of vegetation science

The bases of the French-Swiss school were set down by Braun-Blanquet in his treatise 'Pflanzensoziologie' in 1928, based on work at institutes in Zurich and Montpellier. His phytosociology has found much acceptance throughout the world but by no means all vegetation scientists accepted his system; especially the American and English scientists were sceptical. For a critical review, see Mueller-Dombois & Ellenberg (1974) and Kershaw & Looney (1985).

Basically, there are four steps to the description and classification according to the French-Swiss school (Kershaw & Looney, 1985; Mueller-Dombois & Ellenberg, 1974; Schaminée *et al.*, 1995; Shimwell, 1971; Westhoff & Den Held, 1969).

Surveying. Firstly, the vegetation scientist has to investigate variations in the vegetation of the study area, by walking across the terrain and observing differences in vegetation structure, vegetation height, color, etc..

Sampling. After this survey, locations are carefully chosen where vegetation samples (relevés) are taken. The relevés should together cover all the variations discovered in the surveying stage. Each relevé should meet the following requirements:

- The plant cover should be as homogeneous as possible (no gradients, not dominated by one species in one half of the sample, no large openings, etc.).
- The habitat should be uniform within the plant community, as far as can be determined.
- The relevé should be large enough to contain most of the species belonging to the plant community. In general one uses a standard size that depends on the type of vegetation, e.g. $1-4$ m^2 for pasture lands, $5-10$ m^2 for meadows and $100-200$ m^2 for woodlands (Schaminée *et al.*, 1995).

Then all the species within the relevé are denoted. On a combined scale the number and cover of species are estimated, and also information about the height of the plant cover and the phenological state of the species is gathered.

Classification. The field information is then processed in a table with the relevés as rows, the species as columns and each table element containing information about the number/cover of the species-relevé combination in question. Species that show similar distributions among the relevés are placed together by shifting the rows of the table. Also, by moving columns, those relevés are put side by side in the table that have similar species compositions. In this way a *synthesis table* is created, containing clusters with a characteristic composition of both species and relevés. This table is used for the classification of the vegetation into types.

Relatively small clusters are used for the segregation of *associations*, the basic vegetation units of the French-Swiss school. Each association is typified by *character species* and by *differential species.* A character species is one that shows a distinct maximum concentration in a well-definable association. It may be interpreted as "a key species by which individual communities in the field can be identified as members of a particular community type"(Mueller-Dombois & Ellenberg, 1974). The diagnostic value of differential species is lower. Those diagnostic species show, in comparison to related associations for which they differentiate, only a preference for a certain association.

The same technique used in deriving associations is also used for *alliances*, vegetation units that comprise one or more associations and that are also characterized by both character- and differential species. Alliances in turn, may be grouped into *orders* and orders into *classes*.

The different ranks of vegetation units are usually designated by a particular ending added to the root of the scientific genus name of one or two especially characteristic species (Table C.1).

Mapping. Eventually a vegetation map of the study area can be made, using the character species, differential species and, if desired, the synthesis table to locate the vegetation types distinguished. During this field work, new variations in the vegetation may be discovered, which can lead to new relevés and an extension of the original classification.

In the Netherlands phytosociology according to the French-Swiss school is common practiced. Westhoff & Den-Held (1969) published a description of all vegetation types of the Netherlands, containing per vegetation type information about the diagnostic species, the habitat and the distribution within the Netherlands.

Table C.1: Nomenclature according to the hierarchical phytosociological system of Braun-Blanquet. Examples and codes according to Westhoff & Den Held (1969).

Vegetation rank	Ending	Example	Code
Class	etea	Molinio-Arrhenatheretea	25
Order	etalia	Molinietalia	25A
Alliance	ion	Filipendulion	25Ab
Association	etum	Valeriano-Filipenduletum	25Ab1

This standard work has been used by Dutch vegetation scientists as the main source of reference. Each vegetation type has a code of figures and characters (Table C.1). This code reflects the hierarchial rank of the vegetation type concerned.

Appendix D

Generation of two artificial divisions

Only species involved in the comparison were used to generate both a random division and a geographical optimal division. These amount to 552 species for the comparison with the alliances and 475 for that with the associations. The species for the comparison with the alliances were randomly divided among 41 artificial units creating a *random division*. This number lies right in between the number of ecotope types and alliances being compared. For the comparison with the associations, 43 artificial random units were created.

To obtain the *geographical optimal division*, the same species were first being clustered with UPGMA (Unweighted Pair Group Method using Arithmetic Averages) (Sneath & Sokal, 1973) according to their correlation coefficients (Eq. 3.1). The results of the clustering were presented in a dendrogram, part of which is shown in Fig. D.1 (accompanying codes for ecotope types and alliances are *not* included in the clustering procedure itself; they simply illustrate that kilometer square data provide ecological and phytosociological meaningful clusters).

A division can be generated by drawing - at a certain value of the correlation coefficient - a vertical line through the dendrogram. In this way clusters are split up - the artificial units - in which some species get lost for the division because their cluster becomes too small (< 3 species). To enable comparison between the alliances and associations, a correlation level is sought, the numbers of split up units being near 41 and 43, respectively. On comparison of the alliances, a maximum of 39 units and 355 diagnostic species was reached; with the associations these figures were 39 and 296, respectively. The generated divisions are considered to be optimal because their internal correlations are as high as possible (a division with the lowest possible external correlations has not been made).

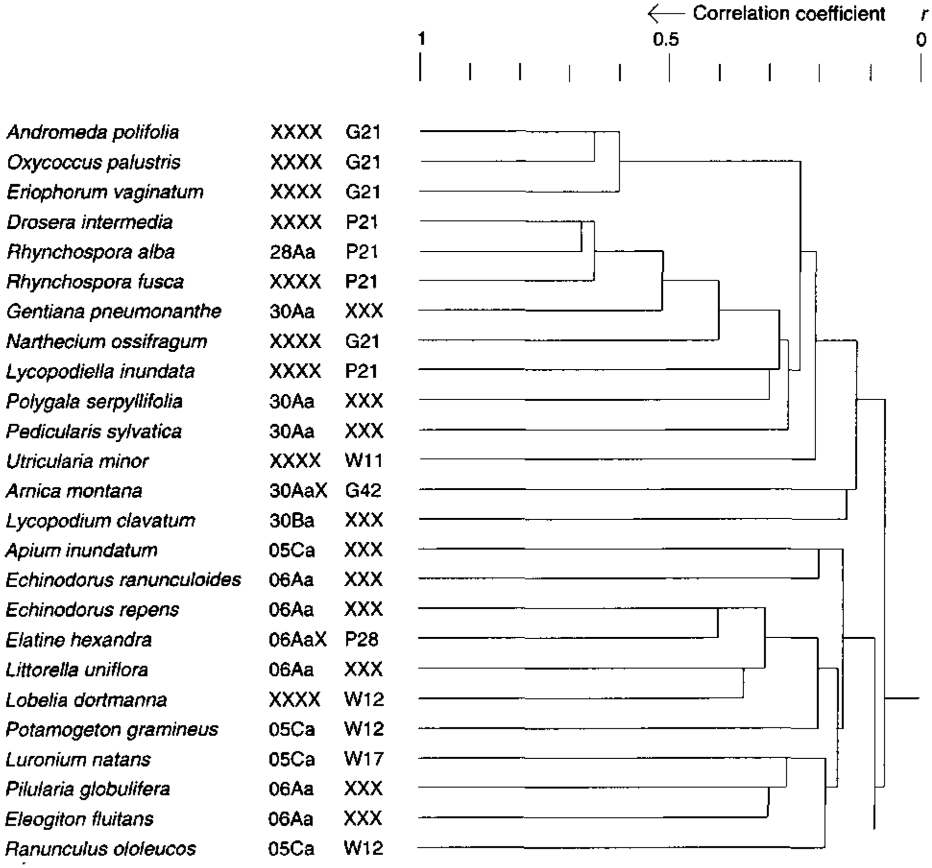


Figure D.1: Part of the dendrogram, generated with the 'Unweighted Pair Group Method using Arithmetic Averages' from FLORBASE.

Appendix E

Derivation of the conservation valuation formula

Derivation of Eq. (5.1)

To obtain conservation values for classification units, such as plant species and vegetation types, we are searching for a positive function $V = \mathrm{f}(A)$ that for every occurrence $A > 0$ meets the following two conditions (corresponding to respectively Assumption 1 and Assumption 2, mentioned in Section 5.2.3):

1. V is a descending function, hence: $V' < 0$
2. AV is an ascending function, hence: $(AV)' > 0$

A function that satisfies both conditions is:

$$V_1 = A^{-c} \quad \text{condition: } 0 < c < 1 \text{ and } A > 0 \tag{E.1}$$

Proof:

ad 1. $V_1' = -cA^{-c-1} \Longrightarrow V_1' < 0 \quad \text{if } c > 0$

ad 2. $(AV_1)' = (-c+1)V_1^{-c} \Longrightarrow (AV_1)' > 0 \quad \text{if } c < 1$

Eq. (E.1) yields values between 0 and 1. In practice, we want V-values between a given minimum V_{min} and a given maximum V_{max}, corresponding to respectively A_{max} and A_{min}. To arrive at a function that satisfies, we first multiply Eq. (E.1) by A^{c}_{max} and obtain:

$$V_2 = \left(\frac{A_{\text{max}}}{A}\right)^c \quad (0 < c < 1 \wedge A > 0) \tag{E.2}$$

It is easy to see that V_2 has a minimum of 1 when $A = A_{\text{max}}$. Because we want the minimum function value to be V_{min} in stead of 1, we multiply Eq. (E.2) by V_{min} and arrive at (with $V = V_{\text{min}}V_2$):

$$V = V_{\text{min}} \left(\frac{A_{\text{max}}}{A}\right)^c \quad (0 < c < 1 \wedge A > 0) \tag{E.3}$$

Finally, we define c in such a way that $V(A_{\text{min}}) = V_{\text{max}}$, or:

$$V_{\text{max}} = V_{\text{min}} \left(\frac{A_{\text{max}}}{A_{\text{min}}} \right)^c \tag{E.4}$$

Hence:

$$c = \log \left(\frac{V_{\text{max}}}{V_{\text{min}}} \right) \left[\log \left(\frac{A_{\text{max}}}{A_{\text{min}}} \right) \right]^{-1} \tag{E.5}$$

The condition that accompanies Eq. (E.3) ($0 < c < 1$) is fulfilled when:

$$\frac{A_{\text{min}}}{A_{\text{max}}} < \frac{V_{\text{min}}}{V_{\text{max}}} < 1 \tag{E.6}$$

In Section 5.2.3 Eqs. (E.3), (E.5) and (E.6) are placed together under Eq. (5.1).

Derivation of Eq. (5.3)

A special case of Eq. (5.1) is the one that meets the following condition (statement of Helliwell, 1973):

$$0.5AV = 0.15A \times V(0.15A) \tag{E.7}$$

Substitution of Eq. (E.3):

$$0.5AV_{\text{min}} \left(\frac{A_{\text{max}}}{A} \right)^c = 0.15AV_{\text{min}} \left(\frac{A_{\text{max}}}{0.15A} \right)^c \tag{E.8}$$

Hence:

$$c = \frac{\log \frac{0.15}{0.5}}{\log 0.15} = 0.63463 \tag{E.9}$$

With this c-value and $Z_{\text{min}} = 1$, Eq. (5.1) is rewritten as Eq. (5.3).

Symbols and specific terms

List of symbols

Symbol	Definition	Dimension	Page
A	Degree of occurrence	-	158
C	Completeness fraction	[-]	104
c^*	Feeding resistance	[T]	39
c_1	Vertical resistance aquitard	[T]	39
c_1'	Vertical resistance topsystem	[T]	39
c_{dran}	Drainage resistance	[T]	39
c_{o}	Ditch resistance	[T]	39
c_{rad}	Radial resistance	[T]	39
D	Density	[-]	156
d	Spatial distinction	[-]	74
Gt	Groundwater table class	[-]	44
GWT	GroundWater Table class	[-]	44
H	Distance between aquitard and p	[L]	39
H_{p}	Phreatic head	[L]	39
kD	Transmissivity of phreatic aquifer	[$\mathrm{L}^2\mathrm{T}^{-1}$]	39
k_{v}	Vertical hydr. conductivity phreatic aquifer	[$\mathrm{L\,T}^{-1}$]	39
M	Average number of hour squares per UFK	[-]	162
N	Groundwater recharge	[$\mathrm{L\,T}^{-1}$]	39
O	Overlap	[-]	88
p	Surface water level	[L]	39
P_{w}	Wetted perimeter	[L]	39
q	Drainage	[$\mathrm{L\,T}^{-1}$]	39
r_{e}	Mean external correlation coefficient	[-]	74
RMS	Root-Mean-Square	-	95
r_{o}	Internal correlation coefficient	[-]	74
r_{s}	Rank correlation coefficient (with ties)	[-]	15
S	Indicator value score	[-]	86
s	Seepage from regional aquifer	[$\mathrm{L\,T}^{-1}$]	39
S_1	Starting (indicator value) score	[-]	112
$S_{0.2}$	0.2%-score	[-]	112
SGL	Spring Groundwater Level	[L]	35, 208
S_{max}	Maximum indicator value score	[-]	110
SSL	Spring Surface water Level	[L]	35
T	Threshold value	[-]	100, 109

Symbol	Definition	Dimension	Page
UFK	UurhokFrequentieKlasse	[-]	12
v	Indicator value	[-]	84
V	Conservation value classification unit	[-]	52, 158
V_{km}	Conservation value km-square	[-]	52, 166
x	Number of aggregated km-squares	[-]	113
α	Num. of common indicator species km-square	[L^{-2}]	97
λ	Leakage factor	[L]	39
φ	Hydraulic head regional aquifer	[L]	39

List of specific terms

Curriculum vitae

Jan-Philip Maria Witte werd geboren op 15 juni 1955 in Amsterdam. Na bestudering van diverse vormen van zowel lager als middelbaar onderwijs, begon hij in 1976 aan een studie aan de Landbouwhogeschool van Wageningen, thans Landbouwuniversiteit Wageningen (LUW). In maart 1983 studeerde hij af in de richting Cultuurtechniek-B.

Daarna volgden vier jaar van kortdurende betrekkingen. Eerst verrichtte hij een jaar vrijwilligerswerk, zowel op het gebied van de ecohydrologie als op het gebied van de informatica. Vervolgens verwierf hij een aanstelling van een half jaar bij het Rijksinstituut voor Natuurbeheer (RIN) in Leersum, waar hij ecohydrologische beschrijvingen van natuurreservaten maakte. In 1985 kreeg hij een betrekking van een jaar bij de provinciale waterstaat van Zeeland in Middelburg. Hij berekende hoeveel zoet water vanuit het Zoommeer naar de Zeeuwse eilanden dient te worden aangevoerd, om verzilting van de landbouwgronden tegen te gaan. In 1986 volgde een periode van een jaar bij de vakgroep Natuurbeheer van de LUW, waar hij hydrologisch onderzoek deed ten behoeve van het ecologisch herstel van de Chaamse beken. Een vaste aanstelling kreeg hij begin 1987 bij het Rijksinstituut voor Integraal Zoetwaterbeheer en Afvalwaterbehandeling (RIZA) van Rijkswaterstaat. Als projectleider was hij verantwoordelijk voor het hydrologisch onderzoek in het proefgebied van de Hupselse beek. Tevens legde hij bij het RIZA de eerste fundamenten voor het ecohydrologische model DEMNAT.

Sinds 1991 werkt hij als universitair docent bij de vakgroep Waterhuishouding van de LUW. Hij verzorgt onderwijs op het gebied van hydrologie, ecohydrologie en - tot voor kort - natuurbouw. Tevens verricht hij ecohydrologisch onderzoek, vaak in opdracht van de Rijksoverheid.