Changing land use
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Changing land use:

a general introduction
Changing land use

Over a period of 4500 years human activities have made their mark on the Dutch landscape. The first farmers cut and burnt the forest to create temporary agricultural fields, which were used for a few years and when the soil became exhausted due to increased leaching after deforestation (Spek 2004), new areas of forest were exploited (shifting cultivation). On the acid, infertile soils dwarf-shrub dominated heather vegetation developed (Gimmingham 1972), preceded by herb-rich grasslands (Waterbolk 1954). Eventually these fields developed into woodland again due to ecological succession.

Over time the human population increased and consequently there was enhanced human impact on the landscape. When settlements became a permanent landscape feature, the land around them was exploited in a rotational system of cropping for short time intervals, followed by long periods of fallow land. Under the influence of livestock grazing these fields maintained an open character (Spek 2004).

In the early Middle Ages (400 A.D.) a new agricultural system developed on the Pleistocene sands (Bieleman 1992). In this system, arable fields were fertilized with nutrients gathered from the adjacent areas of wasteland. The wastelands were utilised by grazing with domestic stock and were also used for sod cutting. The cut sods were used as bedding in the stables and the mixture of sods, animal dung and urine were used to fertilize the arable fields. The regularly flooded, brook valleys were used for hay making to feed the animals during the winter. In this way there was a continuous net transport of nutrients from the wastelands to the arable fields. Only species adapted to nutrient-poor soil conditions survived in the wastelands.

This agricultural system reached its climax at the beginning of the 19th century. At this time the Netherlands were covered with 600,000 to 800,000 ha of heathland (De Smidt 1975). Socio-economic changes and the introduction of artificial fertilizers at the end of the 19th century marked the end of this system. Heathlands were no longer a necessary part of the agricultural cycle. As a consequence of the importation of wool from Australia, the number of sheep grazing on the heathlands decreased substantially (Bieleman 1992). The natural succession of heathland to forest was no longer prevented. There were also opportunities for the conversion of heathland into agricultural land. In addition, large areas of heathland were acquired for afforestation (Brouwer 1968). As a result of these alternative land uses the amount of heathland had dropped dramatically to 100,000 ha by 1940 (De Smidt 1975).

This decline continued until 1961 when reclamation of heathland was prohibited by law, (Anonymous, 1988). However, prohibition of reclamation of heathland to agricultural land was not sufficient to stop the continuing decline. The area of heathland reduced still further due to military land use, intensive recreation and de-
velopment into woodland due to the lack of management. In 1983 no more than 42,000 ha of heathland remained, mainly in small and isolated remnants (Anonymous 1988). This fragmentation of heathland would result in the endemic populations of plant and animal species becoming more vulnerable to extinction (Ouborg 1993; Van Treuren 1993; Boerrigter 1995; De Vries et al. 1996). At the same time, the ecological quality of the heathlands was negatively influenced by the surrounding environmental impacts. The intensive agricultural land use, which developed in the Netherlands after the 2nd World War, resulted in acidification, eutrophication and desiccation of heathland. These impacts resulted in increased dominance of grass species with many characteristic heathland species disappearing or becoming rare (Diemont 1996; Tamis et al. 2005).

At the end of the 20th century, changes in the agricultural policy of the European Union stimulated the transformation of agricultural fields into land with enhanced nature conservation value (De Wit 1988). This new policy provided opportunities to reduce the detrimental environmental influences of the surrounding areas on existing heathlands, and to increase the area of typical heathland communities.

This thesis focuses on the prospects for restoration of heathland communities on former agricultural fields on the sandy soils of the Pleistocene deposits in the Netherlands.

Heathland communities

The term ‘heathland’ refers to a number of phyto-sociological communities dominated by evergreen dwarf shrubs with a well-developed layer of mosses, in which trees and tall shrubs are scarce (Gimmingham 1972; De Smidt 1975). The term heathland is also used to refer to open landscapes characteristic of nutrient-poor soil conditions. In addition to the dwarf-shrub dominated areas, these landscapes include heathland pools, moorland, inland sand dunes and scattered groups of trees. In this thesis I use the term ‘heathland’ in the broad context of open landscapes.

In the slightly undulating part of the Netherlands of Pleistocene origin, dry, humid and wet areas alternate. In places where calcareous groundwater reaches the surface, or loamy layers are present near the surface, slightly nutrient enriched conditions do occur at a local scale. These landscapes include a high diversity of plant communities over relatively small distances, which are characterised by low production (figure 1.1). Communities comprise a highly characteristic flora and fauna.

The acid soils are covered with dwarf-shrub dominated communities. Under dry conditions a vegetation type occurs which is phyto-sociologically assigned to the alliance Calluna-Genistion pilosae (dry heathland). About 23,000 ha of this community remain (Anonymous 1988), and therefore it is still relatively abundant. In
disturbed places, for example along sand roads or the edges of arable fields, the *Thero-Airion* alliance (oligotrophic grassland) may occur (Schaminée *et al.* 1996). This is a pioneer community initially dominated by annuals. Later, perennial species become established, which eventually out-compete the annual species. In the absence of repeated disturbance the community develops into grass-dominated vegetation. Under wet conditions the alliance *Ericion tetralicis* (wet heathland) occurs. It is also found along the edges of acid heathland pools. Only 1,000 ha of this community remain in the Netherlands (Anonymous 1988).

The land area of other, often more species rich, heathland vegetation is even less. These communities are often found only in fine-grained gradients within the heathlands and brook valleys (De Graaf 2000). Under slightly buffered conditions the alliance *Nardo-Galion saxatilis* (matgrass sward) is found. The alliance *Caricion nigrae* (small sedge marsh) occurs under wet, moderately acid conditions (Grootjans *et al.* 1988; Schaminée *et al.* 1995). Under slightly acid to neutral, and moist to wet conditions, the alliance *Junco-Molinion* (fen meadow) occurs. Fen meadows are amongst the most species-rich communities of the Netherlands, which include many endangered species. Only 30 ha of well-developed stands of this community are left (Van Leeuwen 1954; Schaminée *et al.* 1996). In the shallow heathland pools with intermediate pH the alliance *Hydrocotylo-Baldellion* (soft-water communities of shallow lakes) may occur.

Characteristic heathland fauna encompass birds, amphibians, reptiles, butterflies, dragonflies and other invertebrates. Heathland is an important biotope for bird species such as *Tetrao tetrix* (Black grouse), *Caprimulgus europaeus* (European nightjar), *Lanius excubitor* (Great grey shrike), *Saxicola torquata* (Common
stonechat, *Anthus campestris* (Tawny pipit), *Oenanthe oenanthe* (Northern wheatear) and *Lullula arborea* (Wood lark). The heathland habitat supports amphibians and reptiles such as *Rana arvalis* (Moor frog), *Lacerta agilis* (Sand lizard), *Coronella austriaca* (Smooth snake) and *Viper berus* (Adder). Characteristic heathland invertebrate fauna includes; butterflies such as *Maculinea alcon* (Alcon blue), *Plebeius argus* (Silver-studded blue) and *Hipparcia semele* (Grayling), dragonflies such as *Lestes virens* (Small emerald damselfly), *Coenagrion lunulatum* (Irish damselfly) and *Leucorrhiza dubia* (White-faced darter), spiders such as *Eresus cinnaberinus* (Ladybird spider) and *Atypus affinis* (Purse-web spider), carabideae such as *Cicindela campestris* (Green tiger beetle) and *Carabus nitens* and grasshoppers such as *Mertrioptera brachyptera* (Bog bush cricket) and *Dectitus verrucivorus* (Wart biter) (Opdam & Helmrich 1984; Van de Bund 1986; Roos et al. 2000). Characteristic animal species usually are not restricted to a specific plant community, but exploit different parts of the heathland mosaic during different phases of their lifecycle or diurnal patterns of behaviour (Mabelis 1987).

**Restoration**

*Ecological restoration* is defined as ‘the process of assisting the recovery of an ecosystem that has been degraded, damaged or destroyed’ (SER 2002). Two approaches can be distinguished in restoration (Wheeler 1995). The aim of the first approach is to counteract the negative factors causing deterioration in the biological quality of nature reserves where site conditions have not been severely disturbed. The aim of the second approach is the complete reconstruction of communities where site conditions have been severely changed. The latter approach is the subject of this research because the nature conservation interest of intensively exploited fields, in general, is very low and abiotic conditions have been changed considerably.

When this situation occurs the initial requirement is *environmental restoration* (Bakker & Londo 1998). This is ‘a deliberate action to initiate natural abiotic processes and/or the creation of favourable conditions for nature development’. Such measures are initiated by man and generally involve artificial changes to site conditions. Once abiotic conditions have been restored, a *restoration management* can be implemented to stimulate the establishment and development of the desired target communities. Such measures are often long lasting and can vary considerably over time depending on the speed of community assembly. Gradually communities can evolve to the point at which only *maintenance management* is required. The duration of this is unlimited and relatively constant over time.
Top soil removal

Communities with low productivity are characterized by nutrient-poor soil conditions. Due to fertilisation of agricultural land, nutrient concentrations have been substantially increased (Gough & Marrs 1990; Pywell et al. 1994; Aerts et al. 1995). Ecological restoration on agricultural soils therefore first of all must initiate a substantial reduction in soil fertility (Marrs 1993). Traditionally, soil fertility was reduced by undertaking grazing or haymaking (without fertiliser application), but these approaches usually take decades to reduce soil fertility to sufficiently low levels (Bakker 1987; Bakker 1989; Marrs et al. 1998). Recently, removal of top soil has emerged as a technique that will accelerate nutrient impoverishment. Top soil removal involves taking away the upper layers of the soil profile (figure 1.2). Removing soil also removes the nutrients contained within it, and this will result in an immediate reduction in soil fertility (Marrs 1993).

Figure 1.2. Schematic representation of the soil structure of an agricultural field. The arrow indicates the depth for complete top soil removal.

Top soil removal for the purpose of ecological restoration was undertaken some times during the 1970s and 1980s, but use of the technique increased rapidly from the early 1990s onwards (De Ridder 1998; Van Uytvanck & Decler 2004). It has been used most frequently to modify individual fields, but in some projects several tens or hundreds of acres of agricultural fields have been transformed to enhance nature conservation development by removing top soil (De Ridder 1998). Although top soil removal proved to be generally effective in reducing soil fertility, restoration of the desired target community was not always achieved (Aerts et al. 1995; De Ridder et al. 1998; Jansen & Roelofs 1996; Thormann et al. 2003). Only a few cases of top soil removal resulted in the rapid establishment of the desired community (De Ridder et al. 1998; Jansen et al. 2004).
Top soil removal however is a radical and expensive measure. To increase the effectiveness of ‘top soil removal projects’, and to identify locations where successful restoration by means of top soil removal may be possible, detailed insight is necessary about the key ecological processes driving community development and species composition of communities, following deliberate removal of top soil.

**The subject of this thesis**

Local communities assemble from regional species pools through a series of filters or stages, whereby abiotic conditions and biotic interactions act as sieves (Zobel et al. 1998). This thesis focuses on the abiotic conditions and key assembly processes determining community development (figure 1.3).

Restoration of communities first requires that suitable abiotic conditions be restored, followed by immigration of plant propagules or breeding populations of animals into the area and subsequently becoming permanently established (Van...
Diggelen & Marrs 2003). Soil fertility, pH and the groundwater regime are important factors which differentiate communities (De Graaf et al. 1994; Van Diggelen et al. 1997; Bal et al. 2002). A high input of nitrogen from the atmosphere is considered to be an important threat for many communities characteristic of nutrient-poor soils (Fangmeijer et al. 1994; Roelofs et al. 1996; Bobbink et al. 1998).

After appropriate physical and chemical conditions have been created, the characteristic plant and animal species may colonize the restored area. Colonization occurs from source populations, which may occur in the soil seed bank or the immediate surrounding areas. The importance of the soil seed bank is dependant on the longevity of soil seed populations (Thompson et al. 1997), whereas colonization from surrounding areas depends on the presence of source populations and on the dispersal strategy of species and the presence of dispersal vectors (Strykstra et al. 1998; Ozinga et al. 2004).

In the later stages of vegetation development other factors will become important (Pywell et al. 2003), such as interspecific competition and facilitation. The impact of these factors becomes more significant with increasing vegetation cover, height and density. The availability of safe regeneration niches ultimately determines the establishment of colonizing species (Grubb 1977; Tilman 1997; Bakker 2000; Isselstein et al. 2002).

In order to gain further insight into the effectiveness of top soil removal on the restoration of low production plant communities on former agricultural fields, the following questions need to be addressed:

- In which direction and to what extent does top soil removal change site conditions (e.g. pH, nutrient availability, groundwater conditions)?
- What is the quantitative input of atmospheric nitrogen deposition?
- Which species colonize restoration sites and what is the trajectory of community assembly?
- Is there a correlation between abiotic site conditions and subsequent community development?
- Can we observe general trends in the colonization of restored sites by characteristic plant or animal species?
- Is the success of species in colonizing restoration sites related to specific life-history traits?
- How do physical and chemical soil conditions change over time?
- How does species composition of developing communities change over time?
- What are the long-term restoration prospects of low production plant communities on sites where top soil has been removed?
- Which factors contribute to, or constrain, the success of restoration?
- What kind of management should be undertaken to enhance the success of restoration?
Evaluating restoration success

The effectiveness of attempted restoration can only be evaluated when clear targets are set initially (Hobbs & Norton 1996), and then followed by monitoring (Bakker et al. 2000). Often, the aim of restoration is the re-instatement of former characteristic species and typical community diversity (Wheeler 1995). In the present study the targets for restoration are seven characteristic low production plant communities that were once widespread on the Pleistocene sands of the Netherlands. These communities are defined at the level of alliances (see chapter 4), and are further referred to as target alliances. The development of these target alliances is followed by monitoring the occurrence of the characteristic species. These species are further referred to as target species (see chapter 4).

I also pay attention to the development of the heathland fauna following top soil removal. Characteristic species of oligotrophic conditions, that is heathland and inland sand dunes, are set as the target for restoration (see chapter 6).

Study sites

In order to develop generalizations about the potential for restoration of communities’ characteristic of nutrient-poor soils, the research was undertaken at eight sites. These sites, all located on the Pleistocene derived soils present in the north of the Netherlands, represented the entire range of starting conditions found on decalcified, sandy soils, mainly fed by rainwater. The sites are located in the Drenthe district in the north of the Netherlands (figure 1.4). The soil consists of sand, which overlays a thick layer of boulder clay in the shallow subsoil. In a few sites the boulder clay is exposed in some parts to the soil surface (Westhoff & Barkman 1968). This clay layer is almost impervious to vertical water movement (Baaijens et al. 1998). The climate of the northern Netherlands is Boreo-Atlantic. Annual precipitation is about 800 mm yr\(^{-1}\).

Details of the study sites are shown in table 1.1. Before removal of top soil was undertaken, all study sites had been used as pasture or arable fields for a period of 40 years or more. Intensive agricultural exploitation and loading with high amounts of artificial fertilizer started in the Netherlands in the early 1970s. On the arable field in the north of the Netherlands, potato and sugar beet are the dominant crops. Removal of top soil was undertaken between 1989 and 1995, and varied from removal of the entire agriculturally exploited layer (about 30 to 50 cm) down to the mineral sand, to the local removal and re-allocation of the removed soil within the site. Using this methodology, gradients were created from the lower areas where the mineral sand was exposed, to elevated areas where the heavily fertilized layers were accumulated. Following the engineering works, the sites were
grazed by large herbivores fenced contiguously with adjacent nature reserves as a single grazing block.

Table 1.1. Characteristics of the study sites.

<table>
<thead>
<tr>
<th>Site</th>
<th>Size (ha)</th>
<th>Coordinates</th>
<th>Year of reclamation</th>
<th>Former agricult. exploit.</th>
<th>Year of top soil removal</th>
<th>Type of top soil removal</th>
<th>Type of herbivores</th>
</tr>
</thead>
<tbody>
<tr>
<td>Aekingerbroek</td>
<td>20</td>
<td>52°55’N 6°18’E</td>
<td>Before 1920</td>
<td>Pasture</td>
<td>1992</td>
<td>Deep</td>
<td>Cattle + sheep</td>
</tr>
<tr>
<td>Bakkeveensterduinen</td>
<td>3</td>
<td>53°04’N 6°16’E</td>
<td>± 1930</td>
<td>Pasture</td>
<td>1989</td>
<td>Ranging from shallow to deep</td>
<td>Cattle + sheep</td>
</tr>
<tr>
<td>Dellebuursterheide</td>
<td>25</td>
<td>52°57’N 6°08’E</td>
<td>± 1930</td>
<td>Both, but last years pasture</td>
<td>1993</td>
<td>Ranging from shallow to deep</td>
<td>Cattle + horses + sheep</td>
</tr>
<tr>
<td>Eemboerveld</td>
<td>10</td>
<td>53°02’N 7°01’E</td>
<td>1910 – 1930</td>
<td>Arable field</td>
<td>1991</td>
<td>Ranging from shallow to deep</td>
<td>Cattle (only in summer)</td>
</tr>
<tr>
<td>Eexterveld</td>
<td>2</td>
<td>53°00’N 6°42’E</td>
<td>1940 – 1955</td>
<td>Pasture</td>
<td>1994</td>
<td>Ranging from shallow to deep</td>
<td>Cattle + horses</td>
</tr>
<tr>
<td>Ennemaborg</td>
<td>7.5</td>
<td>53°11’N 7°01’E</td>
<td>Before 1900</td>
<td>Arable field</td>
<td>1992</td>
<td>Deep</td>
<td>Horses</td>
</tr>
<tr>
<td>Hullenzand</td>
<td>1.5</td>
<td>52°46’N 6°34’E</td>
<td>1940 – 1955</td>
<td>Arable field</td>
<td>1993</td>
<td>Deep</td>
<td>Cattle + sheep</td>
</tr>
<tr>
<td>Tichelberg</td>
<td>2</td>
<td>53°01’N 7°00’E</td>
<td>Before 1900</td>
<td>Pasture</td>
<td>1992</td>
<td>Deep</td>
<td>Horses</td>
</tr>
</tbody>
</table>
Outline of the thesis

The first part of this thesis deals with the environmental conditions following top soil removal and the subsequent development of the target communities. First, I describe the effects of different depths of top soil removal on soil fertility and relate this to vegetation development and the establishment of target species (chapter 2). In chapter 3 the spatial distribution of atmospheric nitrogen deposition on restored sites is examined in relation to the adjacent agricultural landscape. Nitrogen deposition was measured at different distances to local farmhouses. Chronological development of the species composition of the target alliances is described in chapter 4. Species composition in permanent plots is compared with vegetation descriptions of well-developed reference stands. The success of restoration of the species composition of the communities occurring on fields where top soil was removed, is related to the availability of source populations.

The second part of the thesis focuses on general patterns in the restoration potential of individual plant target species. The species are divided into two groups, either ‘good’ or ‘poor’ colonizers (chapter 5). Life-history traits related to abiotic preferences and dispersal abilities of both groups are compared. Chapter 6 examines general patterns in the colonization by ground dwelling fauna (represented by the carabid beetle group) of sites where top soil was removed. Colonisation success is related to life-history traits of the species, reflecting habitat preference and dispersal ability.

In the final chapter opportunities and constraints for the long-term restoration of low production communities on former agricultural soils are evaluated and discussed. Consequences for the design of the spatial arrangement of nature reserves at a landscape scale and incorporation in to strategic landscape planning are discussed.