Restoration of ditch bank plant diversity : the interaction between spatiotemporal patterns and agri-environmental management

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Chapter 1

General introduction

Biodiversity in agricultural landscapes

One of the most important types of land use in Europe is agriculture. Agricultural landscapes cover over 45% (180 million ha) of the enlarged European Union, with around 103 million ha of arable land, 65 million ha of permanent grassland and 12 million ha of permanent crops (Verburg et al., 2006). Agricultural landscapes offer a wide variety of conditions, due to a combination of natural factors such as soil condition and water availability, and human factors like differences in land use intensity (Donald et al., 2001; Benton et al., 2002). Agricultural landscapes have thus provided unique habitats for many wildlife species and are of great importance to the conservation of biodiversity. Around 50% of all species in Europe depend on agricultural habitats (EEA, 2004).

During the last decades, however, biodiversity losses have occurred in agricultural landscapes at an unprecedented scale. Agricultural practices have shifted from extensive farming systems to either abandonment of farmland or intensification of land use, both of which are considered to be threatening farmland biodiversity (Fig. 1).

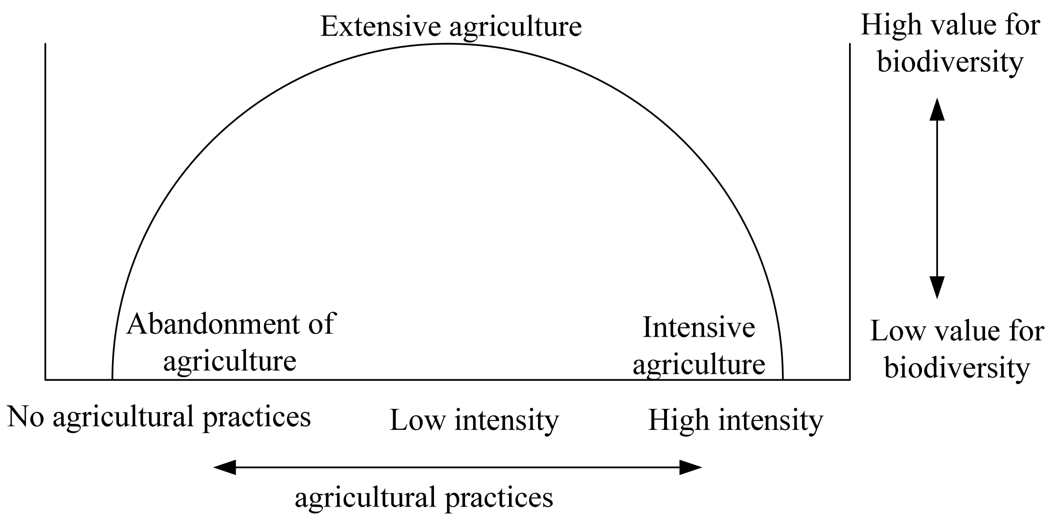


Fig. 1. Relationship between biodiversity in agricultural systems and intensity of agricultural practices (Ostermann, 1998).

Land abandonment is a common phenomenon especially in the regions where agricultural productivity is relatively low (Baldock et al., 1996). The percentage of abandoned arable land in Estonia, for example, was 2% in 1992, and has since dramatically increased to 25% (EEA, 2004). The impact on farmland biodiversity will in most cases be unfavourable (Stoate et al., 2009). At the same time, however, land use intensification is also regarded as one of the most important factors contributing to diversity losses in agricultural areas in Europe (Stoate et al., 2001). Gregory et al. (2000) reported a dramatic decline of bird species in the UK between 1970 and 1998, with Grey partridge (*Perdix perdix*) for instance declining by 82% and Tree sparrow (*Passer montanus*) by 87%. Similar declines in invertebrates and plants have been widely documented in agricultural areas (Petit et al., 2003; Henle et al., 2008).

Land use intensification mainly includes the conversion of complex natural or seminatural ecosystems (grassland) to simplified managed ecosystems (arable fields), and the intensification of resource use, like increasing fertilizer or pesticide input (Tscharntke et al., 2005). Figure 2 shows the effect of landscape complexity on biodiversity at different levels of farming intensity. The biodiversity differences between intensive and extensive farming are most obvious in simple landscapes. Reidsma et al. (2006) assessed land-use intensity change and the related biodiversity loss in the European Union and found that ecosystem quality was lowest in intensively used agricultural areas in lowlands like the Netherlands and northern France.

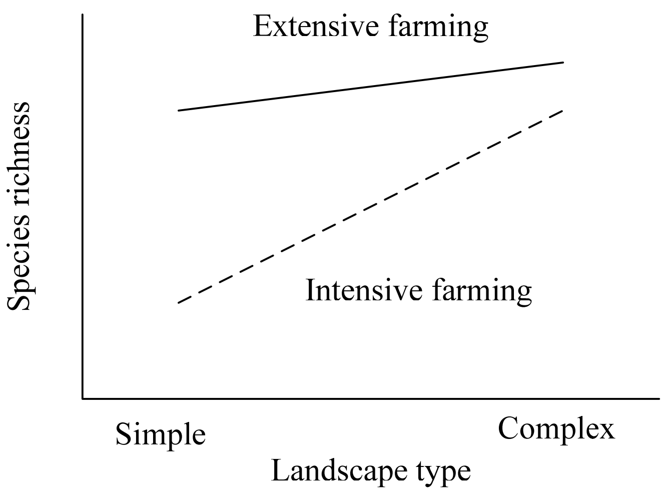


Fig. 2. Diversity of arable land weeds under different management (extensive and intensive) and different types of landscape composition (simple vs. complex) (Roschewitz et al., 2005).

Initiatives to improve biodiversity in agricultural landscape

In order to restore or improve biodiversity, agricultural areas in Europe are now implementing a wide range of strategies at both regional and national levels, including initiatives like the Pan-European Biological and Landscape Diversity Strategy (PEBLDS, 1995), the EU biodiversity action plan for agriculture, and environmental legislation such as the Birds and Habitats directives (79/409/EEC, 92/43/EEC). They mainly focus on either conserving remnants of natural or seminatural areas or enhancing biodiversity in valuable agricultural areas.

Nature protection in the European Union is regulated mainly by the Birds and Habitats directives. It calls for the establishment of a network (Natura 2000) which consists of sites designated under the Habitats directive (Special Areas of Conservation, SACs) and the Birds directive (Special Protection Areas, SPAs). Member States adopt conservation measures on SACs involving appropriate management plans and other measures which correspond to the ecological requirements of the natural habitat types and the species of community interest. SPAs designated under the Birds Directive are managed in accordance with the ecological requirements for bird habitats. The conservation objectives should be met while taking account of economic, social, regional and recreational requirements. It is for the member states to establish the most appropriate methods and instruments to implement the directives and to achieve the conservation objectives for Natura 2000 sites.

Outside protected nature areas, the Common Agricultural Policy (CAP) is the main policy framework affecting conservation of agricultural areas with high ecological value at EU level. Two major relevant elements are agri-environment schemes (AES) and less favoured area payments. AES are considered to be the most important policy instruments to protect biodiversity in agricultural landscapes. They were first introduced by the European Commission (EC), which approved the use of national subsidies for farmers as part of the program. By 1987, countries like England, Germany and the Netherlands had implemented AES. In 1992, the EC adopted the Agri-environmental Regulation EC/2078/92 as part of the Common Agricultural Policy (CAP) reform, and AES became compulsory for member states. Currently, about 25% of all farmland in the fifteen older member states of the EU is covered by some kind of AES (EU, 2005). The main objectives of AES are to counteract the negative effects of modern agriculture on the environment by providing financial incentives to farmers for applying environmentally friendly agricultural practices. Farmers in less favoured areas are eligible for payments per hectare in addition to conventional CAP support, which will generally increase the profitability of farming in marginal areas under natural constraints. As such they are potentially an effective tool for preventing abandonment of ecologically valuable farmland, and may contribute to biodiversity provided they do not create incentives for intensification and particularly overgrazing.

In the Netherlands, the national parliament initiated a new policy called 'Relatienotabeleid' in 1975. One purpose was to establish reserves with optimal conditions for plants and other organisms in agricultural areas. In view of the limited area of nature reserves available in the Netherlands, management contracts with farmers were introduced, in which farmers were paid to provide environmental benefits by applying the following treatments: postponement of mowing and grazing, lower fertilizer input and stocking rates and reducing drainage. The first farmer started to participate in the new 'Relatienotabeleid' schemes in 1981, and the management measures became substantial after 1990 (Beintema et al., 1997). After the Regulation EC/2078/92 was introduced in 1992, postponement of mowing and grazing became the main agri-environmental measure in the Netherlands. In 2000, a new countryside stewardship subsidy scheme named 'Subsidieregeling Agrarisch Natuurbeheer' was introduced. From that time on, farmers implementing ditch bank management were only recommended to use zero fertilizer inputs, low stocking rates, lower ditch cleaning frequencies and extensive mowing and grazing regimes.

Ditch bank vegetation in the Netherlands

In landscapes dominated by agriculture, the former biodiversity is now mostly retained in small-scale landscape elements like ditch banks, field margins and hedgerows (Joenje et al., 1994; Bunce et al., 1998; De Snoo, 1999; Geertsema et al., 2002; Smart et al., 2006). In the Netherlands, ditch banks have a total length of 300,000-400,000 km and are an important feature of the agricultural landscape (Higler, 1994). These ditch banks now function as an important refuge for many formerly common grassland, wetland and hayfield species in terms of survival and diversity (Melman et al., 1991; Blomqvist et al., 2003b). They appear to offer more opportunities to maintain plant diversity, for the following reasons: (1) many ditch banks still harbour species-rich vegetations, including less common species like *Lychnis flos-cuculi* and *Iris pseudacorus*; (2) species-rich ditch banks can be found adjacent to intensively managed fields (Melman et al., 1991; Van Strien, 1991); (3) ditch banks form a economically marginal part of the farm and their grass production is irrelevant on a total farm scale, making it possible to apply extensive management to this habitat.

The peatland areas in the western parts of the Netherlands are among the most intensively exploited areas in Western Europe. The peat bogs that were formed in this area after the last glacial period were later reclaimed and cultivated, causing many changes to the landscape. Long and narrow grassland parcels, separated by shallow ditches or canals, dominate today's reclaimed peat bog landscape and are used for dairy farming. Although species-rich and flower-rich hayfields and pastures were present in these lowlands for centuries, the intensified agricultural activities in recent years have led to the original vegetation being largely replaced by species-poor pastures with a *Poa-Lolietum* vegetation (De Boer et al., 1982; Jansen et al., 1983). The first cause of this is thought to be dairy farming practices, resulting in a rise in nitrogen fertilization from about 70 kg N $ha^{-1}yr^{-1}$ in 1945 to around 250-300 kg N ha^{-1} yr^{-1} in 1980. From the 1980s onwards, however, several of the intensive farming practices, such as fertilizer applications, have been reduced to the 1960s levels. The second potential cause is that land-use has changed from a varied use of fields to their being used as alternate pastures that are often mown early for silage and grazed afterwards. Furthermore, water tables were lowered by drainage to enable intensive grazing and the use of modern, heavy machinery throughout the year.

Although, as mentioned above, the remnants of the grassland communities can still be found on the Dutch ditch banks, the vegetation of these ditch banks is also becoming more and more impoverished. Records over the past 30 years show that the species diversity on ditch banks has been declining (McNeely et al., 1995; Blomqvist et al., 2003b). Many species that until recently were common in the farming landscape, such as *Caltha palustris* and *Lychnis flos-cuculi,* are now receding (Clausman and Groen, 1987).

Conservation strategies on ditch banks: nature reserves and AES

The nature reserves development approach opts for the conservation and restoration of former farming landscapes with their associated extensive forms of agriculture and diversity of wildlife. These reserves harbour a wider range of plant species than the surrounding area (Kremen et al., 2004). However, nature reserves can only cover a limited area. High land prices and conflicting land user interests are major issues, especially in densely populated areas. Although the National Ecological Network (NEN) in the Netherlands was established to expand the total area of nature reserves to protect wildlife habitats, the conservation areas remain so fragmented that the Netherlands will be unable to meet its international obligations on biodiversity conservation (MNP, 2007), suggesting that the effectiveness of nature reserves is rather limited. Maintenance and increase of biodiversity are thus still hampered by the problem that reserves tend to be small and many dispersal processes have been disrupted in today’s increasingly fragmented landscape (Ehrlen et al., 2006; Kiviniemi, 2008).

As reported above, the AES strategy aims to protect the diversity of species and habitats by offering farmers financial incentives to use “nature-friendly” farming practices on certain parts of their lands. Earlier AES used on ditch banks comprised a regime of zero fertilizer inputs, extensive grazing and postponement of initial mowing and grazing at the start of the season. The latest schemes continue to recommend nutrient reduction, but impose few restrictions on the timing of mowing or grazing (DLG, 2000). Although the evaluation of AES has received more attention in recent years, their efficiency in terms of biodiversity conservation is still questioned (Kleijn and van Langevelde, 2006; Blomqvist et al., 2009). Further studies have shown that colonization was a more important factor determining species richness than extinction (Blomqvist et al., 2003b), so management practices such as lower nitrogen levels and postponed mowing, which focused on extinction, failed to prevent diversity loss on ditch banks (Kohler et al., 2008; Blomqvist et al., 2009).

Factors affecting plant diversity on ditch banks

Traditionally, plant diversity was largely attributed to various environmental (biotic and abiotic) factors, such as nutrients, water supply and intensity of disturbance (Ellenberg, 1996). During the last decades, changes in species composition of plant communities as well as the decline and endangerment of numerous plant species were usually interpreted as the result of the decline of environmental quality due to intensification, abandonment or the complete loss of habitats (Condit et al., 2002). Looking at the conservation strategies on ditch banks, we found that many management practices have focused on restoring soil conditions by refraining from applying fertilizers on ditch banks and adapting mowing and grazing regimes. However, these measures do not to seem increase species diversity (Blomqvist et al., 2003b; Kleijn and van Langevelde, 2006). Although restoration of soil conditions is necessary to maintain species diversity, the management approach will still not be effective if seeds are lacking in the soil seed bank or if dispersal from nearby source populations is limited (Bakker and Berendse, 1999).

Because species richness was found to be low and seed bank composition is dissimilar from the vegetation (Bakker and Berendse, 1999; Blomqvist et al., 2003a; Blomqvist et al., 2006), the enhancement of species richness seems to depend on dispersal from species-rich source populations (Crawley and Brown, 1995; Cousins and Lindborg, 2008; Kohler et al., 2008). Although dispersal was not discussed as an important factor in maintaining diversity up until a few decades ago (Fenner, 1985; Murray, 1986), it has attracted growing attention with the increasing fragmentation of habitats in the agricultural landscape. Much recent theory addresses the processes governing diversity in "meta-communities" or networks of local communities connected by dispersal (Hubbell, 2001; Leibold et al., 2004).

There is widespread evidence that dispersal is a controlling factor for the survival of plant communities and, that it therefore limits species richness and diversity (Eriksson, 1998; Cain et al., 2000; Zobel et al., 2000). In agricultural areas, dispersal distances are always limited by spatial configurations like the isolation of habitats and characteristics of the matrix surrounding the habitats (Fleishman et al., 2001). Moreover, most plant species can only actively disperse their seeds over a few metres and are therefore effectively dispersal-limited (Cain et al., 2000). At larger spatial scales, rare long-distance dispersal events are considered an important factor in shaping and maintaining communities (Cain et al., 2000). The seeds, aided by vectors such as water, wind or agricultural activities (Nathan, 2006), have the potential to reach sites that are separated from the source populations by long distances or physical barriers (Levin et al., 2003; Soons and Bullock, 2008). It therefore became increasingly obvious that processes and vectors combined with different land-use practices are the key to the dispersal capability of plants. Another important factor is assumed to be the distance between seed source populations and target areas, due to the limited dispersal capacity of most plant species (Fenner, 1985).

Objectives and outline of this thesis

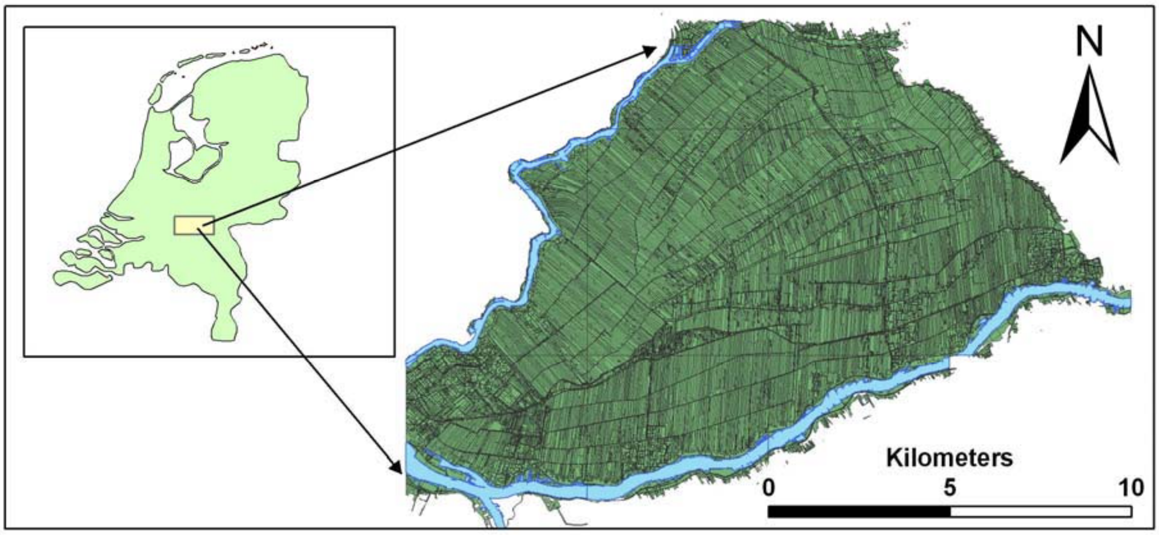
So far, studies of plant diversity on ditch banks have primarily focused either on the effects of ecological mechanisms on individual species (Blomqvist et al., 2003a; Blomqvist et al., 2003b) or on direct management and species richness (Melman et al., 1991; Van Strien, 1991). Initiatives to improve the biodiversity, however, were not as successful as expected (Kleijn and Van Langevelde, 2006; Blomqvist et al., 2009). Effective protection of plant diversity requires more detailed knowledge of ecological mechanisms, especially at larger scale, of plant communities and possible management practices. This study therefore focused on two objectives.

The first objective was to assess *the spatial and temporal patterns of species diversity and the relevant factors on ditch banks*. In recent years, conservation biology has matured and its emphasis has shifted from the management of individual species within habitats to the preservation of entire communities (Whitfield, 2002; Tuomisto et al., 2003). This paradigm shift has required considerable attention to be given to the way patterns of biodiversity vary across spatial and temporal scales. Regional diversity patterns are a result of local processes, underlying environmental heterogeneity and species dispersal among local communities (Collins et al., 2002). Beta diversity, which is the difference in species composition between local communities, is a major determinant of species diversity at regional scale and can be used to measure how variation among local communities contributes to regional diversity (Margules and Pressey, 2000; Ferrier, 2002). We therefore examined how patterns of species diversity change in space and time, as well as the scale dependence of factors that contribute to diversity. Furthermore, since species diversity was poorly protected under management schemes like AES, the selection of additional sites for conservation should be guided by a greater understanding of the species diversity patterns on ditch banks.

The second objective was to explore *possible management for plant diversity restoration based on ecological mechanisms*. Both ecological and economic obstacles often interfere with the creation and maintenance of nature reserves, while the ecological efficacy of agri-environment schemes (AES) is still questioned. A conservation strategy involving integration of nature reserves and agriculture (through AES) has been suggested to improve plant diversity (Steffan-Dewenter and Tscharntke, 1999; Ockinger and Smith, 2007; Cousins and Lindborg, 2008) and we tested whether this strategy can be used on ditch banks. Furthermore, mowing is common practice in grasslands used for dairy farming. In low-intensity farming, it is considered a traditional practice likely to lead to high plant species richness (Huhta and Rautio, 1998). In high-intensity farming, however, it may be regarded as a form of disturbance hampering seed setting in plants. Scientific knowledge about the impact of mowing on seed availability at locations and for dispersal is thus necessary and might help to establish the most effective mowing regime to protect and increase plant diversity.

Research area

Both research questions were addressed by means of analyses of existing data as well as a field study. Our study area encompassed ditch banks in the Krimpenerwaard area, located in the Western Peat District in the Netherlands (51°53’N - 52°01’N and 4°35’E - 4°51’E) (Fig. 3a, 3b). This area can be characterized as a typical Dutch polder landscape and is among the most intensively exploited areas in Europe.



(a)

(b)

Fig. 3. Location of landscapes studied at Krimpenerwaard (**a** and **b**).

The landscape originated about 6000 years B.C. and was formed as a wadden area by the flooding of the lower parts of the Netherlands after the last glacial period. After this area was shut off from the sea by coastal barrier deposits, it transformed into peat bogs. The soil type of the area nowadays consists of peat, while near the rivers, it is bordered by zones of clay and clay-on-peat at greater distances. The current polder land was formed by reclamation of the peat area about 1000 years ago. A network of parallel drainage ditches was created, usually perpendicular to the rivers, resulting in a landscape with long, narrow fields and farmsteads usually near the rivers (Van Strien et al., 1989). Initially, the most distant fields were used extensively, whereas the fields behind the farmhouses near the rivers were used more intensively. As a result of the agricultural intensification process, however, most fields are nowadays exploited intensively (De Boer et al., 1982). The original vegetation types have largely disappeared from the landscape due to the steady lowering of water levels and higher fertilization inputs. The remaining original vegetation types are currently exclusively found on ditch banks and thus serve as a refuge for much of the former biodiversity. The study mainly focused on 25 target species of nature conservation (Appendix in *Chapter 4*). These species were selected because they are not only deemed to be valuable ditch bank plants in Dutch government policy but are also used as criteria for rewarding farmers who implement AES.

In pursuit of the two objectives of this thesis, a series of studies was carried out. The studies relating to the first objective (Part I) are discussed in *Chapter 2* and *Chapter 3*, while those relating to the second one (Part II) are reported in *Chapters 4*, *5* and *6* (Fig. 4).

Part I Spatial and temporal patterns of species diversity

Chapter 2

The spatial and temporal patterns of plant diversity on ditch banks under different types of management were examined by means of additive partitioning of diversity as well as analyses of similarity. First, the relative contribution of diversity components to total diversity was estimated for all species, and a similarity index (Jaccard) was calculated for the pattern of species diversity differences in space. Second, we tested whether these patterns differed between all species and the target species. Finally, we studied whether the patterns of target species respond differently between ditch banks in nature reserves and those in agricultural areas.

Chapter 3

Whereas the previous chapter evaluated relative contributions to diversity at different spatial and temporal scales, *Chapter 3* explains spatial patterns of species composition by taking into account the combined effects of dispersal and environmental factors, using multiple regression on distance matrices (MRM). The vegetation data on ditch banks were used to investigate whether and to what extent the species similarity between plots can be explained by the environmental and dispersal factors. Furthermore, the pattern for the target species was also tested and compared with that for all species. Finally, we focused on the patterns for species with different dispersal strategies.

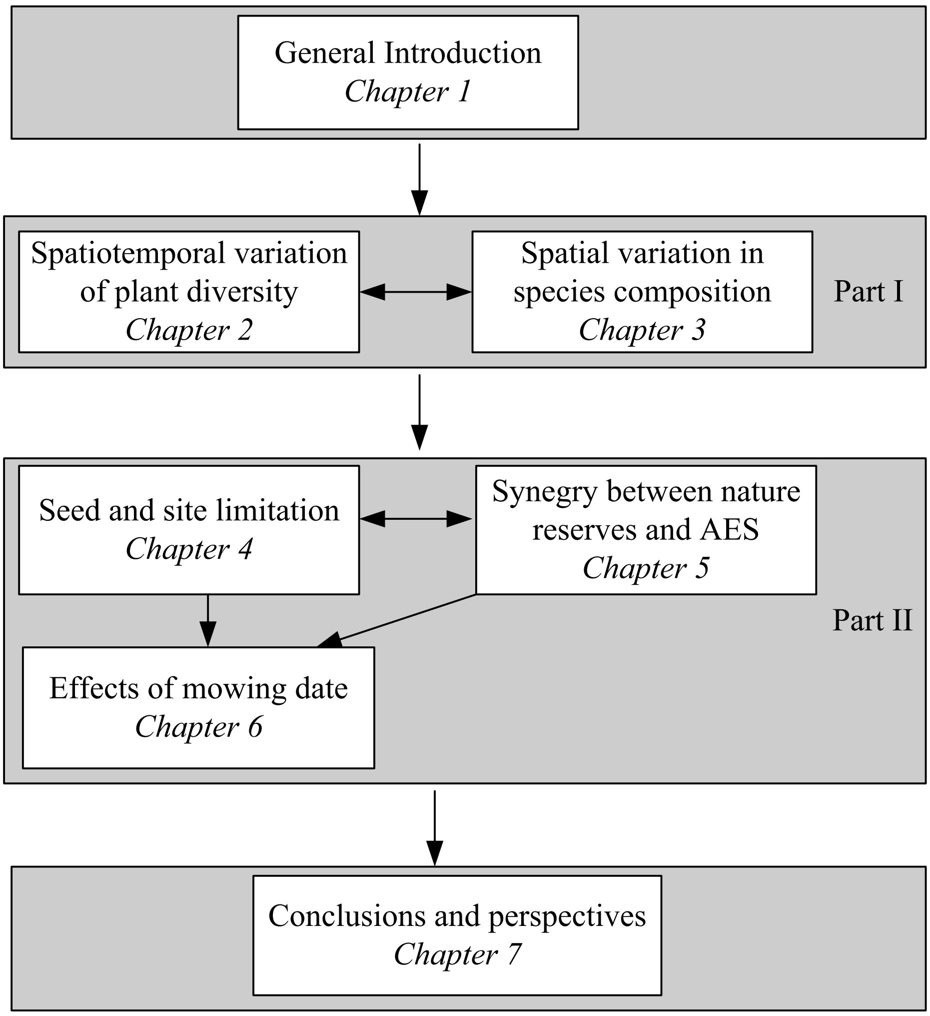


Fig. 4. Structure of the thesis and relation between chapters

Part II Possible managements for plant diversity restoration

Chapter 4

A conceptual model was developed for plant species of ditch banks, to distinguish between site limitation (environmental factors) and seed dispersal limitations.

Studying the restoration sites near species-rich source habitats (nature reserves) enabled us to explore whether dispersal is important to plant diversity restoration. We first tested whether the proximity of nature reserves can improve plant species diversity on the surrounding ditch banks, and then examined whether plant species diversity is higher in ditch banks managed under agri-environment schemes. This was followed by an investigation of the interaction between the presence of nature reserves and AES areas.

Chapter 5

The study reported on in *Chapter 4* thus evaluated the importance of nature reserves for the plant diversity influenced by AES along ditches running transversely from the nature reserve to the farmland. This left unanswered the question of trends in plant diversity along banks running in other directions, to yield an overall picture of how to arrange the nature reserves and AES at the landscape level. This chapter focuses on the effects of the synergy between nature reserves and AES on plant species across a network of ditch banks. We first studied the pattern of plant diversity on successive ditch banks running parallel to a nature reserve, and then made a comparison of the pattern between ditch banks running transverse and parallel to a nature reserve. Finally, we focused on AES and investigated whether ditch banks managed under an AES showed different plant diversity patterns in two directions relative to nature reserves.

Chapter 6

Preliminary studies (*Chapter 4* and *Chapter 5*) revealed that conservation management does not increase connectivity by decreasing seed limitations for plant species. Since mowing might be a measure to increase seed dispersal, we undertook a comprehensive study of the effect of variations in mowing date on seed availability for seed transportation on ditch banks under four different management regimes (nature reserves, AES with long-term management, AES with short-term management and conventional management). Two research questions were addressed, one to check whether the seed-setting of ditch bank plant species is affected by the timing of mowing, the other to assess whether this effect varies with different management regimes.

Chapter 7

This chapter briefly summarizes and discusses the results of the previous chapters. It also proposes guidelines for ditch bank plant diversity conservation and options for future research.

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