THE ECOLOGY AND PSYCHOLOGY OF AGRI-ENVIRONMENT SCHEMES

William F.A. van Dijk

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The ecology and psychology of agri-environment schemes

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Contents

1. General Introduction

1. Agri-environment schemes

Over the last decades there has been an alarming worldwide decline in biodiversity (Balmford et al. 2005, Butchart et al. 2010). Europe has committed itself to halt the loss of biodiversity by 2020 (EEA 2006, Maes et al. 2012). Of all species in Europe, 50% is estimated to be associated with farmland, including many threatened species (Kristensen 2003, Kleijn et al. 2011). Over the last 50 years the biodiversity related to farmland has declined greatly due to intensification of the agricultural practice and the abandonment of species rich areas under agricultural management (Stoate et al. 2001, Stoate et al. 2009). The decline in farmland biodiversity has been particularly steep in the lowlands of north-west Europe such as southern England, northern France, northern Italy, Belgium, Germany and the Netherlands due to the strong agricultural intensification in these areas (EEA 2009). This tendency is not expected to halt in the upcoming decades (Tilman et al. 2001).

To conserve the biodiversity in agricultural landscapes the Europe-wide Natura 2000 network of nature reserves have been established aimed at protecting natural and semi-natural areas with exceptional biodiversity values. However, in order to protect sufficient area to stop the ongoing biodiversity decline, integration of agriculture within EU nature conservation policy is inevitable (Siebert et al. 2006, Balmford et al. 2012).

The most important instrument aimed at conserving biodiversity at productive farmland is Agri-environment schemes (AES) (Burton and Wilson 2006). AES provide financial support€ to help farmers manage their land in a more environmental-friendly way. The objective of the AES is twofold: first they increase the area of land under some form of nature-friendly management to conserve biodiversity. Second, AES may connect fragmented nature reserves as ecological corridors in the landscape (van Dorp et al. 1997, Geertsema et al. 2002, Donald and Evans 2006).

The Netherlands were the first European country to implement AES after the approval of the Policy Document on the Relation between Nature and Agriculture in 1975 (Beintema et al. 1997, Berendse et al. 2004). The UK and Germany followed in 1987 and Denmark, France, Ireland, Italy and Luxembourg in 1992 (Primdahl et al. 2003). After the reform of the Common Agricultural Policy (CAP) in 1992 the implementation of AES became compulsory for all member states under the Agri-Environment Regulation (Council Regulation 2078/92). The costs for implementation were partly financed from the EU budget (50% or 75%, depending on the specific objectives) and partly from the member states (Primdahl et al. 2003, Berendse et al. 2004). The initial uptake of AES among farmers in the member states varied greatly with percentages of over 75% in Austria and Finland, while less than 7% of farmers in Belgium, Greece, Spain, Italy and The Netherlands performed AES(Berendse et al. 2004). Nowadays, the European wide annual expenditure on AES are estimated at € € 5 billion, approximating the costs of the Natura 2000 Network (Gantioler et al. 2010). Similar schemes have been advocated or are already implemented in other parts of the world, such as the United States, Australia and Brazil (Curtis and De Lacy 1998, Attwood et al. 2009, Shiki 2010).

However, research has shown that AES in Europe vary widely in their effectiveness to actually conserve biodiversity (Peach et al. 2001, Berendse et al. 2004, Feehan et al. 2005, Kleijn et al. 2006, Wilson et al. 2007, Kleijn et al. 2011). One of the key factors limiting the outcome of AES is that the prescriptions of current AES are often too shallow to establish a change in biodiversity (Kleijn et al. 2001, Blomqvist et al. 2009). Secondly, often there is no landscape approach applied (Kleijn et al. 2011, Tscharntke et al. 2012) meaning that AES are being performed in unsuitable areas and not nested within the configuration of existing nature reserves and AES (Kuiper et al. 2013). Due to their profound prescriptions and ineffectiveness in altering biodiversity even after 10 years the objective of AES has even been alleged to be back-door production subsidies rather than conserving biodiversity (Burton et al. 2008).

So far, improving current AES has mainly been regarded as an ecological challenge (Kleijn et al. 2011, Uthes and Matzdorf 2013). However, there is a growing awareness among the conservation community that the incorporation of other disciplines such as sociology and psychology is imminent in order to enhance biodiversity conservation, particularly AES (Burton 2004, Balmford and Cowling 2006, de Snoo et al. 2012a). For example, in order to improve current AES, policy makes depend on the willingness of farmers in two ways; (A) farmers in ecological favourable locations must be willing to participate in AES and (B) farmers most be motivated to perform more elaborate measures. In this perspective are farmers one of the most influential groups on the planet on biodiversity given the areas of land they manage. Nonetheless, the body of scientific literature on attitudes, acceptance and uptake of AES by farmers is still meagre compared to the research on effects of AES on biodiversity (Uthes and Matzdorf 2013). In Chapter 6 I will integrate these factors into an interdisciplinary model for ditch bank management.

2. Ditch bank management

One of the most commonly applied types of AES in Europe is the management of ditch banks. Ditch bank management is performed in countries such as the Netherlands, Denmark, Ireland and the UK (Melman 1991, Feehan et al. 2005, Donald and Evans 2006, McCracken et al. 2012, Stutter et al. 2012). Ditch banks are a particularly suitable landscape element in agricultural landscapes to address with AES, because they are known to be relatively species-rich even when adjacent to intensively managed fields (Melman 1991, van Strien 1991). Ditch banks can harbour up to 96% of the total number of species of entire agricultural fields, which often consist of previously common plant species from hay meadows and wet grasslands. (Kleijn et al. 2001, Herzon and Helenius 2008).

Furthermore, ditches are one of the few still common linear landscape elements even in intensive agricultural landscapes. For example, in the Netherlands ditch banks stretch up to 300.000 to 400.000 km (Higler 1994). This makes ditch banks suitable to create an extensive network of corridors within a landscape connecting fragmented nature reserves. Besides conserving the biodiversity in the ditch bank itself, ditch banks can also serve as buffer strips to trap nutrients from adjacent fields to waterways (Stutter et al. 2012).

Finally, farmers generally consider ditch banks as an economically marginal part for the whole farm production. This makes it more likely that farmers are willing to take up measures aiming at lowering the productivity in ditch banks than on the more productive areas of the farm (Kruk et al. 1988, van Strien and ter Keurs 1988, van Strien 1991).

3. Ditch bank management in the Netherlands as a case study for AES

The earliest form of ditch bank management started in the Netherlands in the early 90’s on a small scale due to a lack of interest from farmers to conserve species-rich plant communities (van Strien and ter Keurs 1988). From 2000 on under the new Agri-environmental program Subsidiestelsel agrarisch natuur- en landschapsbeheer the number of participants increased substantially. Prescriptions of management encompassed no manure or ditch slurry in the outermost metre of ditch bank (DLG 2000). These prescriptions were based on previous research that demonstrated that plant diversity on ditch banks can be enhanced by lowering the nutrient input (van Strien et al. 1989, van Strien et al. 1991, Berendse et al. 1992, Bakker and Berendse 1999).

The organisation of Dutch ditch bank management was quite different compared to the majority of AES in Europe. Ditch bank management was performed in collectives, the so called Agri-Environment Cooperatives (EC) (Glasbergen 2000). The EC coordinated the performance of ditch bank management by individual farmers and assisted with the application for subsidy. To motivate farmers to participate in ditch bank management the EC applied a payment by results approach. The collective governmental subsidies from all members were reimbursed to farmers based on the presence of a number of target species in their ditch banks (van Dijk et al. 2013a). The monitoring of target species in ditch banks by farmers may have increased the visibility of the outcomes of the management for farmers, possibly affecting their motivation (van Strien et al. 1988, Clausman 1996, Musters et al. 2001).

This makes Dutch ditch bank management quite deviating from the majority of AES in Europe which usually apply individual management agreements which prescribe certain measures to be performed, which are assumed to have some ecological effect (Burton and Paragahawewa 2011, Burton and Schwarz 2013, Uthes and Matzdorf 2013). Albeit the Dutch ditch bank management still prescribes certain measures, due to the payment by results approach farmers are expected to perform additional measures to increase the number of target species, which would result in a higher reimbursement (Musters et al. 2001, Burton and Schwarz 2013).

4. Effectiveness of Dutch ditch bank management

Over the last 10 years more and more authors have started to question the efficacy of ditch bank management in achieving an increase in the number of plant species in ditch banks. Field studies suggest that there is very little to no increase in the number of species in ditch banks with AES (Kleijn et al. 2001, Blomqvist et al. 2009, Leng et al. 2011). However, most of this research is based on short-term pair-wise comparisons of plots with and without AES. This approach has two drawbacks. First, it is questionable if this pairwise comparison is a methodologically sound comparison. Whereas the control plots are randomly selected by the researcher, the result-oriented approach of Dutch ditch bank management may motivate farmers to selectively apply AES in their most species rich ditch banks, possibly leading to a positive selection effect on the number of plant species (Musters et al. 2001, Leng et al. 2010b). Second, changes in the vegetation over long time periods are not considered. So far there has been only one study that investigated trends in AES and non-AES ditch banks over a 10 year period (Blomqvist et al. 2009), but this study encompassed only six dairy farms. In order to investigate the effects of ditch bank management on the vegetation large scale, long term studies are required, which are lacking so far (de Snoo et al. 2012a, McConkey et al. 2012, Tscharntke et al. 2012). Furthermore, to avoid a selection effect both plots with and without AES should be selected by farmers.

5. *Research questions*

In this thesis I will address the question why Dutch ditch bank management has not been effective in increasing the number of species in ditch banks, and how can this AES be improved. To answer this question I will use long-term data on the plant species composition of ditch banks. These data derivate from the payment by results setup that was applied in Dutch ditch bank management between 2000 and 2009. In order to apply the payment by results system, the EC yearly distributed a number of 100m x 1m quadrats over participating farmers ditches based on the kilometres of ditch bank a farmer managed. The numbers of target species in these quadrats were monitored by the farmers and subsequently checked by the EC. These data provide a long-term dataset (between 2000 and 2009) at a large spatial scale (comprising more than 3900 kilometres of monitored ditch banks managed by 490 individual farmers) on the effects of ditch bank management in the Netherlands. Drawback of this dataset is that there are no monitoring data of farmers without AES. As control group I will use quadrats of farmers in the year that they started the AES, which is not necessarily 2000. This approach avoids the selection effect of farmers, because all quadrats are farmer-selected.

5.1. Chapter 2

In Chapter 2 I will assess trends in the number of target species in ditch banks between 2000 and 2009 based on these data. Furthermore I will characterize the vegetation based on underlying ecological principles. Previous research has demonstrated that the limiting factors for the outcome of ditch bank management can be divided into two categories; site- and seed-limitations (Blomqvist et al. 2006). Seed limitations encompass the availability of seeds in ditch banks by means of the soil seed bank and the dispersal of seeds from nearby sources. Site limitations involve factors at the location of the plant that may inhibit the establishment and survival of the species such as competition for light due to high nutrient content of the soil.

5.1.1. Site-limitations

The ditch bank prescriptions of excluding fertiliser and ditch sludge from the banks mainly aim at reducing competition for light in ditch banks, thus addressing site limitations. Previous research has demonstrated that plant diversity on ditch banks can be enhanced by decreasing nutrient supply (van Strien et al. 1989, van Strien 1991, Berendse et al. 1992, van der Linden and de Jong 1994, Bakker and Berendse 1999). However, small-scale field studies so far have questioned the effectiveness of the AES in actually decreasing nutrient supply (Kleijn et al. 2001, Kleijn et al. 2004, Blomqvist et al. 2009). To investigate the effects of site-limitation I will calculate the temporal trend in the nitrogen demand of the vegetation using the Ellenberg N values of the target species.

5.1.2. Seed-limitations

Previous research has demonstrated that the soil seed bank in Dutch ditch banks is strongly impoverished and provides little opportunity for the restoration of biodiversity in these ditch banks (Blomqvist et al. 2003a). However, in a favourable landscape configuration, seed dispersal from nearby species-rich sources has been demonstrated to alter species richness in ditch banks (Leng et al. 2009, 2010b), although the effect of these sources on nearby ditch banks is strongly dependent on the dispersal capacity of a certain species (van Dorp et al. 1997, Leng et al. 2010a). To investigate the effect of dispersal on the vegetation composition, I will characterize the target species based on their dispersal vector and analyse temporal trends in these different dispersal categories.

5.2. Chapter 3

In chapter 3 I will elaborate further on the effect of landscape configuration on seed-limitations by analysing spatial and temporal trends of the species composition of ditch banks at increasing distance from nature reserves. I hypothesize that nature reserves can serve as species-rich sources of dispersal in the landscape and ditch banks may function as corridors (Leng et al. 2009, 2010b). I will also investigate if the relationship between distance from a nature reserve and species richness will change in time due to colonisation of ditch banks at increasing distance of nature reserves. Again I will calculate these trends for different dispersal categories, to investigate the effect of dispersal vector on the suitability of ditch banks as corridors.

5.3. Psychological factors

The results of chapter 2 may lead to recommendations on how to improve current ditch bank management by extending management prescriptions, while chapter 3 may shed a light on how the configuration of AES in the landscape may be improved. However, in order to have farmers implement more elaborate measures policy makers are dependent on farmers' willingness to take up these measures. Furthermore, a regional body will be required to coordinate the landscape approach on a regional scale and to motivate farmers at suitable locations for these more complex measures (Beedell and Rehman 2000). ECs, which have been coordinating the performance of ditch bank management, may provide such a regional body that can assist with the performance of the AES, but can also help to convince farmers to perform certain measures. There are two ways how more elaborate measures can be implemented: by changing the prescriptions of subsidized management or by trying to motivate farmers to perform additional unsubsidized measures in their ditch banks. These two ways to alter farmers' behaviour lead to the questions investigated in chapter 4 and 5.

5.4. Chapter 4

In chapter 4 I will investigate what psychological variables affect farmers' intention to perform subsidized ditch bank management. To do so I will use an adapted and extended version of the Theory of Planned Behaviour (TPB;(Ajzen 1991)) as a framework. This theory has been proven to be a structured, flexible model that can explain the cognitions that underlie individual farmers' willingness to perform AES (Burton 2004, Fielding et al. 2005, Sutherland 2010, Wauters et al. 2010, Lokhorst et al. 2011). Previous research has demonstrated that attitude towards AES is the main driver of farmers' willingness to perform AES (Lokhorst et al. 2011). However, the collective performance of ditch bank management in ECs may facilitate and create group pressure for farmers to participate in AES. To capture the influence of the EC I will include two new variables. First, the variable group facilitation is added to measure how facilitation by the EC contributes to farmers' intention to perform AES. Second, I will add the variable group norm to capture the influence that peer pressure from fellow EC members performs on farmers. I will compare the results of ditch bank management with another popular AES in the Netherlands; the protection of meadow birds. This way I will be able to compare motivational differences between AES. In this chapter I will enquire specifically about the intention for subsidized ditch bank management, with prescribed measures. Therefore the results obtained in this chapter can be used to assess what variables should be addressed to motivate farmers to perform more elaborate subsidized measures.

5.5. Chapter 5

Previous research has demonstrated that farmers also perform additional unsubsidized measures such as the maintenance of hedges or truncated trees on their farm. It has been demonstrated that variables affecting the intention of farmers to perform unsubsidized measures are different from those that affect subsidized management (Lokhorst et al. 2011). Therefore I will investigate in chapter 5 what motivational factors explain farmers' willingness to perform additional unsubsidized measures. The outcomes of this chapter can shed light on the variables that can increase the willingness of farmers to extend their ditch bank management without receiving subsidy.

5.6. Chapter 6

In chapter 6 we will summarize and integrate the outcomes of the previous 4 chapters. Furthermore we will analyse for a number of measures that farmers take in ditch banks whether they contribute to the restoration of plant species richness in order to develop clear recommendations for ditch bank management. Finally, we will combine the data on target species richness from chapter 2 with the motivational aspects from chapter 4 to create an interdisciplinary model which describes the pathway from motivation to number of plant species in ditch banks. This model may elucidate the relationship between human behaviour and ecological results of conservation policy- which has been advocated in the literature, but has rarely been reported so far (Balmford and Cowling 2006, de Snoo et al. 2012a, Uthes and Matzdorf 2013).

2. Temporal effects of agri-environment schemes on ditch bank plant species

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Abstract
Many of the Agri-environment schemes (AES) implemented in the Western Peat District of the Netherlands have as their objective the conservation of the diversity of ditch bank plants. We investigated the effects of AES on ditch bank species in this area, using a dataset collected by 377 farmers who managed and monitored ditch banks during a 10-year period. We found that species richness has increased minimally over the last ten years in ditch banks. Yet, we found no differences in increases in time between ditch banks with and without AES. In both ditch bank types plant species composition changed to species with higher nitrogen tolerance. Furthermore, species that disperse over long distances by water increased, whereas species with no capacity to disperse over long distances declined in both ditch bank types. This indicates that changes in vegetation composition in ditch banks are affected by other factors than AES.

1. Introduction

As a result of the unremitting intensification of agriculture, the biodiversity on farmland in Europe has declined dramatically in recent decades (Stoate et al. 2001, Strijker 2005, Tscharntke et al. 2005, Stoate et al. 2009, Geiger et al. 2010). Agri-environmental schemes (AES) have been implemented throughout Europe under the Common Agricultural Policy (CAP) to halt this decline and achieve the 2010 biodiversity target (Balmford et al. 2005). For some time, however, the effectiveness of these schemes has been questioned (Kleijn et al. 2001, Kleijn and Sutherland 2003, Berendse et al. 2004, Critchley et al. 2004).

In various countries, such as the Netherlands, Ireland and the UK, ditch banks in agricultural landscapes are under some kind of AES (Feehan et al. 2005, Cole et al. 2012, Stutter et al. 2012). Even today, ditch banks are known to be relatively species-rich (Herzon and Helenius 2008); they can account for up to 96% of the total species richness of entire fields (Kleijn et al. 2001). Furthermore, ditch banks can serve as buffers to trap nutrients from adjacent fields (Stutter et al. 2012).

Nowadays, the Netherlands have one of the most intensive agricultural systems of Europe (LEI, 2010). The intensity of the farming has greatly affected the botanical diversity of fields and neighbouring ditch banks, since the high soil fertility favours a low number of fast-growing species (Blomqvist et al. 2003a, de Snoo et al. 2012b). To conserve the specific ditch bank flora, the Dutch government has selected target species such as Caltha palustris, Silene flos-cuculi and Lotus pedunculatus (van Harmelen et al. 1997), on which AES focuses. These species can be easily recognized and their presence is correlated with a high species richness of ditch banks (Jansen et al. 1989, Kaiser et al. 2010). These species used to be common in damp hay meadows but their numbers have declined sharply in recent decades (Schaminee et al. 2010).

From 1998 onwards Environmental Cooperatives (ECs) coordinated the AES in the Netherlands (Glasbergen 2000). The ECs, who consist of farmers and local citizens, applied for participation on behalf of their members, and also decided on the remuneration. The remuneration for ditch bank management was usually based on a result-oriented principle, according to which, farmers with more target plant species were paid more. This system aimed to motivate farmers to fine-tune their nature management to seasonal and local conditions (Musters et al. 2001). The presence of the target species was usually monitored by the farmers themselves and subsequently verified in the field by the ECs.

AES prescriptions to conserve ditch bank plant diversity comprise excluding fertiliser, manure and ditch sludge from the banks, as experiments have shown that plant diversity on ditch banks can be enhanced by decreasing nutrient supply (van Strien et al. 1989, van Strien 1991, Berendse et al. 1992, van der Linden and de Jong 1994, Bakker and Berendse 1999). However, small-scale field studies have cast doubt on the effectiveness of the AES in decreasing nutrient supply (Kleijn et al. 2001, Kleijn et al. 2004, Blomqvist et al. 2009). Another factor that may hamper the results of ditch bank management is the lack of seed sources of the target species. Previous research has shown that restoration of species diversity from the soil seed bank is very limited in ditch banks (Blomqvist et al. 2003a) as is dispersal from species-rich areas in agricultural landscapes (Blomqvist et al. 2003b, Ozinga et al. 2009). In the vicinity of nature reserves, however, the species richness of the reserves has been demonstrated to influence the surrounding ditch banks, increasing their number of species, even if they are intensively managed (Kohler et al. 2008, Leng et al. 2009, 2010b).

Most of the studies done so far on the effects of AES have been pairwise comparisons between farms with and without AES (Kleijn et al. 2001, Kleijn et al. 2004). The result-oriented principle of the AES, however, is likely to motivate farmers to perform AES in their most species-rich ditch banks (Matzdorf et al. 2008, Blomqvist et al. 2009, Leng 2010). This may affect the reliability of pairwise comparisons. Also, it does not take potential differences in temporal trends in vegetation composition on AES and non-AES ditch banks into consideration, because usually no measurement at the start of the management is performed. The only study on results of AES ditch bank management so far that took these points into account, was relatively small (only six farms)(Blomqvist et al. 2009). Our study is, as far as we know, the first long-term analysis of the results of ditch bank management using a large database. We attempted to answer the following two questions: 1. Do AES in ditch banks lead to a change over time in species composition and the number of target species? 2. Which functional groups respond to the management? We define functional groups based on two mechanisms; The nitrogen demand of the target species present by means of Ellenberg N and differences in dispersal ability among target species.

2. Material & Methods

2.1. Study area

The study area is situated in the Western Peat District of the Netherlands in the provinces of Utrecht, South-Holland and North-Holland, (N 51°53’-52°20’ and E 4°24’-5°18’). The Western Peat District is a rural area. The major land use is agriculture, but there are some small nature reserves with national and international ecological value, in which the vegetation consists predominantly of damp meadow and marsh species (Schaminee et al. 2010). The predominant agricultural activity is livestock farming, particularly dairy farming.

The long narrow meadows used for this purpose are separated by ditches in which the water levels are controlled by the water board (usually 0-40 centimetres below the surface of the field). They normally fluctuate by 10-15 cm over the year (Blomqvist et al. 2003a). The ditch banks are dominated by wet grassland species. The AES with a botanical focus that is most commonly implemented in this area deals with the first metre of ditch bank field margin of pastures counting from the edge of the ditch. The prescribed management is non-application of artificial fertiliser, organic manure or dredged sludge on the ditch bank; mowing and grazing by cattle is allowed (DLG 2000).

2.2. Data collection

We obtained yearly monitoring data collated by 5 ECs in the research area (Hollandse Venen, Vechtvallei, Weide & Waterpracht, Weidehof Krimpenerwaard, and Wijk & Wouden). The data had been collected by 377 farmers, who monitored their ditch banks between June and August in the years 1998-2009. Farmers recorded the presence of target species in a 100 metre long and 1 metre wide quadrat per kilometre of AES-managed ditch bank. Collected data were verified in the field by the EC.

2.3. Data management

The lists of monitored target species per EC contained between 19 and 31 species, and differed among both ECs and years. For a general analysis of target species numbers in time among all ECs we included 10 target species that were monitored in all the

Table 1. Number of quadrats monitored on ditch banks in five Environmental Cooperatives (EC) at different durations of AES management.

	Management duration (years)									
EC	1	2	3	4	5	6	7	8	9	10
Hollandse Venen	233	233	155	127	81	13				
Vechtvallei	927	927	872	802	55	430	384	374	351	300
Weide & Waterpracht	122	65	61	26	22	22	19	19	19	19
Wijk & Wou-den	184	184	136	83	83	74	35	35	34	
Weidehof Krimpener-waard	1054	1054	800	571	369	291	175	175	170	153
Total	2520	2463	2024	1609	610	830	613	603	574	472

years by all ECs (Table 1, see Appendix 1). It was not possible to trace back individual quadrats in time for all ECs. Therefore, we calculated the average number of these 10 target species per quadrat for each farmer in each year. The farmers sometimes changed the number of kilometres of ditch bank they managed, hence changing the number of quadrats monitored. To ensure that the same ditch banks were studied over time, we only analysed the data per farmer from the years in which the number of quadrats was equal to the number in the year a farmer started management. Farmers were not allowed to rotate management over different ditch banks. We assume that in these cases the same ditch banks were monitored. This selection resulted in different maximum durations of management for farmers, with a maximum of 10 years (Table 1). Duration of management is the time in years since a farmer started management. In one EC (Vechtvallei) no monitoring data could be obtained for the year 2004.

Table 2. Number of quadrats monitored per year in Weidehof Krimpenerwaard.

	2000	2001	2002	2003	2004	2005	2006	2007	2008	2009
AES quadrats	153	153	153	153	153	153	153	153	153	153
Control quadrats	710	266	127	43	274	64	81	8	40	

To compare between quadrats with and without AES we performed an in depth analysis on the data from EC Weidehof Krimpenerwaard. We selected Weidehof Krimpenerwaard for our in depth analysis, because this was the only EC that had substantial amounts of farmers starting AES over a wide range of years (Table 2). For this EC it was possible to trace back individual quadrats in time. Weidehof Krimpenerwaard monitored 25 target species (see Appendix 1). Although, only a limited number of species were used in our analyses, previous research has shown that the presence of, for example, Caltha palustris, Lotus pedunculatus and Silene flos-cuculi strongly correlates with total species number, level of extensive use of grasslands and the presence of Red List species (Kaiser et al. 2010).

We analysed numbers present of these 25 target species, instead of the 10 species used in the first analysis. As AES quadrats we selected 153 quadrats among 20 farms that were managed continuously from 2000 until 2009. Farms were selected in such a way that quadrats were more or less evenly distributed over the entire EC. As control plots we used all the quadrats of farms in the EC in the year that they started AES (Table 2), which is not necessarily 2000. We excluded the first year observations of the AES quadrats in the control group to ensure independence between the control and AES quadrats. Former research pointed out that comparing AES quadrats with control quadrats may be biased by a selection effect of farmers performing AES in their most species-rich ditch banks. Control quadrats, on the other hand were selected by researchers, and did not have this bias (Blomqvist et al. 2009, Leng et al. 2009). By using both farmer selected AES and control plots we exclude this bias.

To investigate what ecological mechanisms underlie observed changes in species composition, we rated the 25 target species' long-distance dispersal capacity using Hodgson et al. (1995). In this rating we distinguished between capacity of species to disperse over long distances by means of water, wind or animals. If a species had no specialized morphological features for any of the previously mentioned dispersal vectors it was rated as having 'no-long distance dispersal capacity'. Species that disperse over short distances by ants and species that shed their seeds from a capsule held above the vegetation were considered as well as having no-long distance dispersal capacity. We analysed numbers present of every dispersal category (see Appendix 1). To investigate the nitrogen demand of the target species of the ditch banks we obtained Ellenberg N-values for each target species and calculated the average Ellenberg N value of a quadrat based on the target species present (Ellenberg et al. 1992).

2.4. Statistical analysis

All statistical analyses were performed using Predictive Analytics Software (PASW) (Version 17.0.03; SPSS/IBM Inc., Somers, NY). We analysed the average number of 10 target species per quadrat in all ECs, using a linear mixed model (Laird and Ware 1982). Duration of management was added as a covariate to the model. We nested farm within EC and added both to the model as random factors, to enable the modelling of variation between ECs and farms. To account for variation in the number of quadrats per farm in a given year, this number was used as a residual weighting factor.

For the in depth analysis of the numbers present of 25 target species in Weidehof Krimpenerwaard we performed a Generalized Linear Mixed Model (GLMM) (Bolker et al. 2009) with a Poisson distribution and a log-link. We could trace back individual quadrats in time, therefore we analysed the number of 25 target species per quadrat as the dependent variable. We added farm as random factor. We distinguished between quadrats that were managed continuously between 2000-2009 and grouped them as AES quadrats. Quadrats that were in their first year of management were grouped as control quadrats. Year was added as covariate to the model and management (AES or control), and the interaction year*management as fixed factors. We subsequently reduced the number of variables in the model based on their significance (type III sum of squares), until we obtained the simplest model with the lowest Akaikes Information Criterion (AIC) and significant terms only. Our experimental setup leads to measurements in various years in the ditch banks in the AES group. In the control quadrats we have one-time measurements, namely in the year that AES started. By adding farm as the subject we compensate for variation in farms among years. We did not use a repeated measures design, because quadrats were redistributed yearly over the ditch banks. For the analysis of trends in the Ellenberg N-value we performed a GLMM with a normal distribution and the same procedure as the previously mentioned GLMM.

3. Results

3.1. General analysis

We found an average number of 2.91±1.24 out of 10 target species per ditch averaged over all ECs. The number of these target species increased by 4.4% over 10 years ($F_{1,1240.6}$=4.162, P<0.01) (Fig. 1). The initial number of target species varied greatly between farms (Wald Z: 11.167, P<0.001). We found no significant effect of EC.

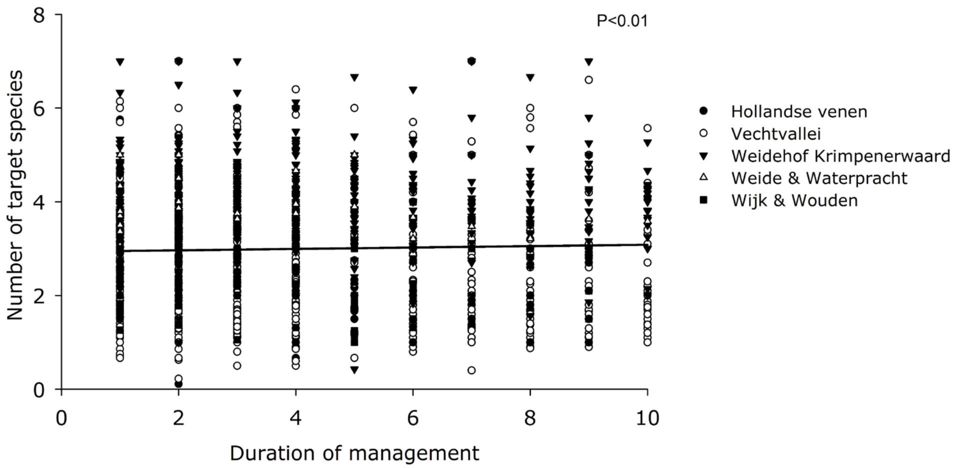

Figure 1. Changes over time in average number of ten selected target species monitored at five Environmental Cooperatives (EC). The line shows the predicted values obtained from a linear mixed model with duration of management as covariate and farm nested within EC, which were added as random factors. Significance level of the line is shown.

3.2. In-depth analysis of Weidehof Krimpenerwaard

The results of the in-depth analysis of Weidehof Krimpenerwaard showed that there is a significant increase of 1.4% yearly over 10 years ($F_{1,1341}$: 14.780, P<0.001) in the number present of the 25 target species monitored in this EC. Yet, this increase did not differ between AES ditch banks and control ditch banks. We neither found a difference in the number of target species present between ditch banks with and without AES. Analyses of different dispersal categories revealed that wind species increased annually by 4.7% ($F_{1,1341}$: 7.652, P<0.01) in AES and control ditch banks, whereas animal-dispersed species showed no temporal trends . However, these two categories included only 1 and 2 target species, respectively. Water-dispersed species (9 species monitored) increased yearly by 2.5% ($F_{1,1341}$: 25.971, P<0.001). This increase did not differ between AES and control ditch banks.

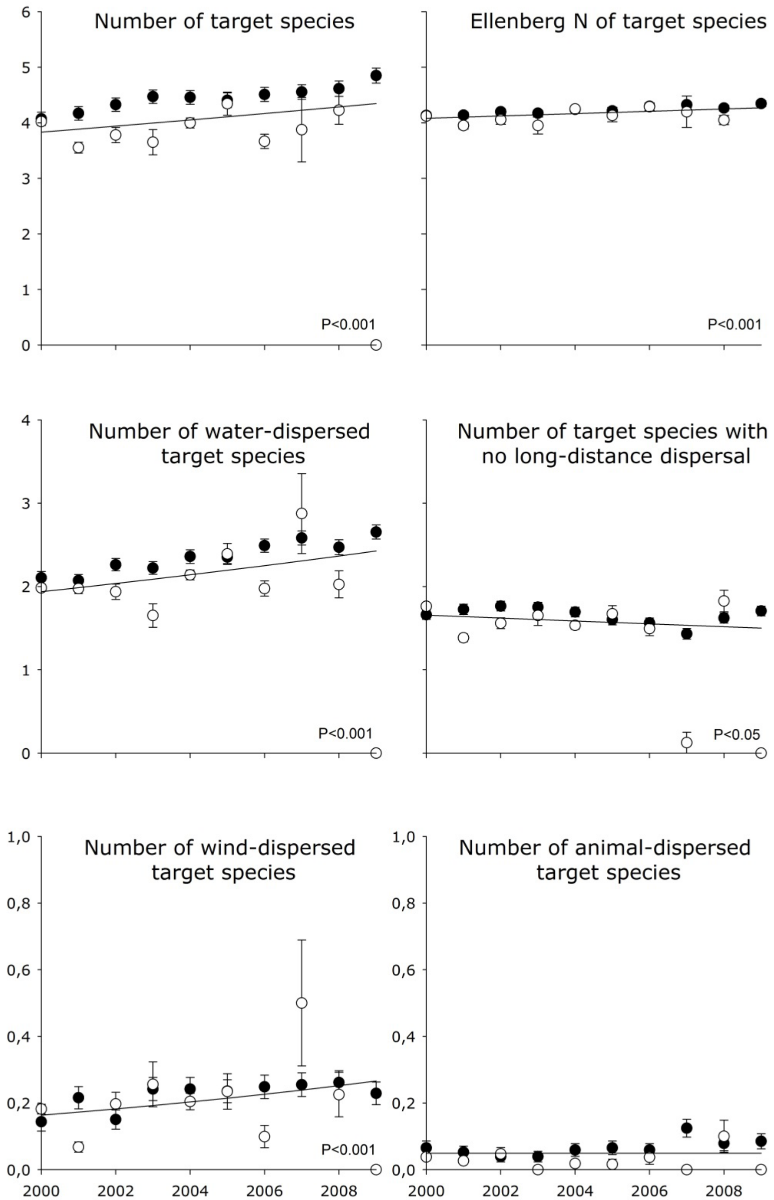

Figure 2. Changes over time in average numbers of 25 target species monitored on AES ditch banks and control ditch banks at Weidehof Krimpenerwaard. Black circles represent the average of AES quadrats with standard error, open circles control quadrats. The line represents the predicted values over time determined by performing a GLMM analysis with AES, control quadrats and their interaction and subsequently removing factors until the simplest model was obtained with significant factors only. Significance level of the line is shown . Since no significant interaction or difference between AES and control quadrats was obtained, only a single line is shown.

Species with no capacity to disperse over long distances (8 species) declined yearly by 1.1% ($F_{1,1341}$: 3.867, P<0.05). This trend was equal in AES quadrats and control quadrats. We found that the average Ellenberg N number for the species present has increased yearly by 0.021 over the last ten years in both AES and control quadrats ($F_{1,1341}$: 15.905, P<0.001).

4. Discussion

Our study demonstrates that the number of target species has shown an negligible increase (1.1% per 10 years). This increase is equal among ditch banks with and without AES. This result corroborates the results of a previous small-scale study on ditch bank management. However, our finding that there is no difference in the number of target species at the start of management between AES and control quadrats, contradicts previous research in which AES quadrats had higher numbers of target species at the start of management (Blomqvist et al. 2009, Leng et al. 2009). However, in those publications the control quadrats were located in researcher-selected ditch banks.

In our experimental setup both the control and the AES quadrats were farmer-selected. This suggests that farmers do indeed perform ditch bank management in their most species-rich ditch banks (Musters et al. 2001, Matzdorf et al. 2008).

The variation in number of target species we found among ECs may be due to variation in soils among ECs (Kleijn et al. 2004) or hydrological conditions (van Strien et al. 1989, Beintema et al. 1997). Furthermore the landscape lay-out of a ditch bank may play a role, affecting seed-dispersal from nearby species-rich sources such as nature reserves (Blomqvist et al. 2003a, Donald and Evans 2006, Leng et al. 2009, 2010b). There are substantial differences in the area of nature reserves among different ECs. 8.8% of EC Weidehof Krimpenerwaard is covered by nature reserves, whereas in EC Wijk & Wouden nature reserves covered only 0.02% of the total area.

We found that particularly water-dispersed species were increasing, but there was no significant difference in the increase in ditch banks that had been under continuous AES management for 10 years compared to the ones that were not. Species with no capacity to disperse over long distances declined significantly between 2000 and 2009, here we found no difference between AES ditch banks and the control ditch banks either. Thus, our results suggest that although AES are not capable of altering species richness, ditch banks themselves are able to serve as corridors in the

agricultural matrix, but their suitability as such differs among dispersal groups. This may be due to the relatively small area that a ditch bank covers (1 m wide). It would be worthwhile to investigate for wind- and animal-dispersed species if wider extensively managed linear elements are able to sustain populations (van Dorp et al. 1997).

A drawback of the dataset that we used in this article is the small number of target species that it covered and the negligible increase that we found in the total target species numbers and that of certain dispersal categories. However, within the categories we found common trends. For example, the general increase in target species numbers in ditch banks was mainly driven by the species Iris pseudacorus, Mentha aquatica and Lythrum salicaria. All these species are water-dispersed species and have a relatively high Ellenberg N value. On the other hand, the negative trend in species with no long-distance dispersal is mainly driven by Myosotis spp. and Silene flos-cuculi, which both showed declining trends in AES ditch banks (-1.5% yearly, $F_{1,379}$: 22.907, P<0.001 and -0.9% yearly, $F_{1,379}$: 12.119, P<0.001, respectively). Although this decline may be driven by only two target species, Myosotis spp. was the second most common target species in the ditch banks (83% of the ditch banks contained Myosotis spp.).

Our results demonstrate that the species composition changed towards species with higher N demand, but there was no difference in the trends between ditch banks with and without AES. This suggests that current AES management is not able to decrease N availability (Blomqvist et al. 2009), while this is actually the main aim of the management prescriptions.

The lack of efficacy of the implemented AES to increase traditional hay meadow species is probably due to the small 'ecological contrast' that these measures provide (Kleijn et al. 2011); excluding fertilization 1 metre out of the ditch bank, while whole-field fertilization is not adjusted provides little difference compared to no management. This was reflected by the non-significant difference between increase of target species in ditch banks with and without management and in the continuing increase of nitrophilous target species.

This demonstrates that conservation measures performed in intensively farmed areas have little effect on the plant diversity. AES in such areas are very costly - the remuneration to farmers was up to €31.25 per target species in a kilometre of ditch bank managed – while the benefits are small (Kleijn et al. 2009).

Besides focussing on decreasing productivity, more attention should be paid to measures aiming at increasing seed dispersal and establishment. Previous studies have demonstrated the crucial importance of seed dispersal (Ozinga et al. 2009), especially within the agricultural landscape (Blomqvist et al. 2000, Blomqvist et al. 2003b, Leng et al. 2009, 2010a, b).

Given these limitations, it is difficult to provide an explanation for the overall increasing trend in species numbers. A recent study of agricultural grasslands in the same area showed declining species numbers, most likely due to the intensification of agricultural practices, until the 1990s, after which the decline appeared to halt (de Snoo et al 2012). However, it is not unlikely that the expansion of plant species adapted to intense agricultural practices is still ongoing. Recent results from a long-term fertilization experiment in grassland showed that species richness declined with fertilization for more than 25 years, but then started to recover as species adapted to the fertilized conditions established in the experimental plots (Pierik et al. 2011). This explanation appears to be supported by our findings of increasing Ellenberg N value of the target species over time, and the importance of dispersal strategies in explaining the temporal trends of different target species.

5. Conclusions and management implications

We conclude that numbers of target plant species in ditch banks slowly increased. However, AES did not have a positive effect on target plant species numbers in ditch banks. The increase in target species can mainly be attributed to water dispersed nitrophilous species. This indicates that current management, which prescribes lowering nutrient input in the outermost metre of the ditch bank, does not succeed to alter the vegetation towards less nitrogen-demanding species. More intensive measures are required to lower nutrient input. However, our results also indicate that linear landscape elements, such

Acknowledgments

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Appendix 1 List of target species. Species printed in bold were monitored for all Environmental Cooperatives (ECs) and included in the first analysis. All species together comprise the target species monitored in Weidehof Krimpenerwaard. Columns Wind, Water, Animal and No Long Distance Dispersal (LDD) show a species' capacity to use the mentioned vector for long distance dispersal. Ellenberg Nitrogen mentions the Ellenberg N value sensu Ellenberg (1992).

	Wind LDD	Water LDD	Animal LDD	No LDD	Ellenberg Nitrogen
Caltha palustris	0	1	0	0	
Cirsium palustris	1	0	0	0	3
Galium palustre	0	1	0	0	4
Hydrocotyle vulgaris	0	1	0	0	2
Lotus pedunculatus	0	0	0	1	4
Silene flos-cuculi	0	0	0	1	
Mentha spp.[a]	0	1	0	0	5
Myosotis spp.[b]	0	0	0	1	6
Ranunculus flammula	0	1	0	0	2
Veronica beccabunga	0	0	1	0	6
Achillea ptarmica	0	0	0	1	2
Centaurea jacea					
Cirsium dissectum					2
Filipendula ulmaria	0	1	0	0	4
Iris pseudacorus	0	1	0	0	7
Lathyrus pratensis	0	0	0	1	6
Leucanthemum vulgare	0	0	0	1	3
Lysimachia thyrsiflora					
Lysimachia vulgaris	0	1	0	0	
Lythrum salicaria	0	1	0	0	
Pedicularis palustris					2
Potentilla palustris	0	0	0	1	2
Prunella vulgaris	0	0	1	0	
Rhinanthus angustifolius					2
Vicia cracca	0	0	0	1	
Total	1	9	2	8	

Lumped taxa: a Mentha aquatica and Mentha arvensis, b Myosotis scorpioides subsp. scorpioides and Myosotis laxa subsp. cespitosa.

3. The effectiveness of ditch banks as dispersal corridor for plants in agricultural landscapes depends on species' dispersal traits

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Abstract
The effectiveness of agri-environment schemes (AES) in enhancing biodiversity in agricultural landscapes is still strongly debated. In the Netherlands, one of the most widely implemented AES is the management of ditch banks to enhance plant species diversity. Previous research has shown that this type of AES has not led to increases in plant diversity. However, this work also showed that the success of this type of AES may depend on the presence of source populations in the surrounding areas. In this study we investigated if species-rich nature reserves can act as seed sources for agricultural ditch banks under AES and whether this function of nature reserves differs among plant species with different dispersal capacities. We used data collected by farmers over a 10 year period to analyse trends in species richness of target plants and in different dispersal groups in ditch banks under AES at different distances from nature reserves.

Our results demonstrate that nature reserves can act as species rich sources in agricultural landscapes and that adjacent AES ditch banks can facilitate the colonization of the surrounding agricultural landscape. However, the suitability of ditch banks as corridors depends on the dispersal capacity of a species. Particularly water-dispersed species clearly spread from nature reserves into the surrounding agricultural landscape along ditches. In contrast, species without adaptations to disperse over long distances do not show these spatiotemporal patterns.

1. Introduction

For decades, biodiversity has been declining in agricultural areas worldwide due to intensification of agriculture and the abandonment of species-rich areas under extensive agricultural management (Foley et al. 2005, Strijker 2005, Stoate et al. 2009). This trend is not expected to halt in the upcoming decades (Tilman et al. 2001), despite international agreements like the Convention on Biological Diversity (CBD), which attempts to enhance global sustainable development in farming systems (Balmford et al. 2005). In Europe, the decline in biodiversity on farmland has been particularly strong in countries in the lowlands of north-western Europe, such as The Netherlands (Donald et al. 2001, EEA 2009, Stoate et al. 2009, Geiger et al. 2010). Measures implemented in Europe to halt the decline in biodiversity on farmland include the creation of nature reserves in which extensive agriculture takes place to conserve biodiversity, and agri-environment schemes (AES), which are designed to restore biodiversity on farmland. These schemes serve two important functions in the protection of biodiversity. First, they increase the area of land under management to conserve biodiversity. Second, land under AES may act as corridors or stepping stones in the landscape to connect fragmented nature reserves (van Dorp et al. 1997, Geertsema et al. 2002, Donald and Evans 2006, de Snoo et al. 2012b). However, AES in Europe vary widely in their effectiveness in actually increasing biodiversity (Peach et al. 2001, Berendse et al. 2004, Feehan et al. 2005, Kleijn et al. 2006, Wilson et al. 2007, Kleijn et al. 2011, Kuiper et al. 2013). In the Netherlands, a substantial part of AES is focused on plant-species richness in ditch banks. Over the last decades, most plant species of former species-rich meadows have been driven back to field margins and extensively managed nature reserves (Kleijn et al. 2001, Geertsema et al. 2002){Schaminee, 2010 #192;van Strien, 1988 #272}. AES focused on these plant species include the management of the outermost metre of ditch bank of pastures, where no fertilizer or ditch sludge may be deposited (DLG 2000). The most important indicator to monitor the effect of this AES is the richness of plant species of ditch banks.

Previous research in the Netherlands has shown that although AES ditch banks on average contained more species than non-AES ditch banks, no further increase in the number of species in AES ditch banks was found after longer periods of AES (Kleijn et al. 2001, Kleijn et al. 2004, Blomqvist et al. 2009, van Dijk et al. 2013a). Explanations for these disappointing results include soil nutrient status that did not reach sufficiently low levels, high productivity and competition for light (van Dijk et al. 2013b) and limited recruitment from the seed bank (Blomqvist et al. 2003a). Another factor may limit the restoration of species richness in many ecosystems is seed dispersal (Blomqvist et al. 2003b, Ozinga et al. 2005, 2009).

Nature reserves in agricultural areas generally contain a larger number of plant species than the agricultural fields (de Snoo et al. 2012b). These areas may act as a source of species that disperse into the surrounding agricultural landscape (Kohler et al. 2008, Leng et al. 2009, 2010b). AES ditch banks may then act as a corridor by which species from nature reserves can disperse throughout the landscape (van Dorp et al. 1997, Geertsema et al. 2002, Donald and Evans 2006, Soomers et al. 2010).

Previous research found that plant species with different dispersal strategies differ in the distances from the nature reserves at which they can establish along ditch banks (van Dorp et al. 1997, Leng et al. 2010a). Long-term studies are required to investigate the effects of dispersal on vegetation composition in AES ditch banks, but so far these studies are lacking (de Snoo et al. 2012b, McConkey et al. 2012, Tscharntke et al. 2012).

Here, we investigated the changes in plant species richness in ditch banks under AES and neighbouring nature reserves over the last 10 years. We investigated whether plant species richness in ditch banks decreased with increasing distances from the nature reserves. Next, we tested if ditch banks can serve as a corridors by comparing the relationship between species richness and distance to a reserve in different time periods. Finally, we compared these patterns among groups of species with different dispersal stategies.

2. Material & Methods

2.1. Study area

Our research was conducted in the Krimpenerwaard area (N 52°00'-51°53' and E 4°33'-4°50') (Fig. 1). It covers 14908 hectares and is part of the Western Peat District in the Netherlands. The most common agricultural land use in this area is pasture used for grazing or hay meadow for silage production for sheep and dairy cattle. Fields in this area are typically long (500-900 m) and narrow (30-60 m). Fields are separated by one to four meter wide ditches, which have a combined length of 3927 km in this area. The water level in the ditches is controlled by the water board and varies between 0 and 50 cm below the surface of the field within a year (Blomqvist et al. 2003a). The main soil type in the area is peat, with clay soils near the rivers.

2.2. AES quadrats

Ditch bank management in the Netherlands is usually implemented under the supervision of an Environmental Cooperative (EC) (Glasbergen 2000), also known as agri-environmental collectives (Verhulst et al. 2007) or farmers' collectives (Leng et al. 2010b). These bodies, which usually consist of farmers and local citizens, apply for AES on behalf of the farmer. Some of the ECs apply a remuneration of their members on the base of achieved results, i.e. the presence of selected plant species. This approach aims at an increased efficacy of AES by motivating the farmer to pay attention to the results of his management (van Strien et al. 1988). Furthermore, a farmer is more likely to implement management in the parts of his land, where adapted management is expected to be most efficient (Musters et al. 2001, Matzdorf et al. 2008, Burton and Schwarz 2013).

We obtained data of 1494 quadrats in ditch banks with AES management, which were managed and monitored by 63 farmers. AES were established in 2000 and these plots were monitored in the years 2000, 2005 and 2009. AES management encompassed no application of fertiliser, manure or dredged sludge on the first metre of ditch bank, measured from the water's edge; mowing and grazing by cattle is allowed (Melman 1991, DLG 2000). Quadrats were 100 m long and 1 m wide and were redistributed yearly by the EC over the ditch banks managed. The number of quadrats per farmer is proportional to the number of kilometres of ditch banks managed by that farmer.

3. The target plants were selected by the Dutch government as species that are correlated with plant species richness along ditch banks and easily recognisable (Jansen et al. 1989). In total, 25 plant species were monitored by farmers (Supplementary Table A1). In 2000, when management started, farmers were made familiar with recognizing the target plant species. In the following years, observations of at least 25% of the quadrats per farmer were verified by specialists. To make the comparison between quadrats per farmer among years as reliable as possible, we included only ditch banks that were managed in all three

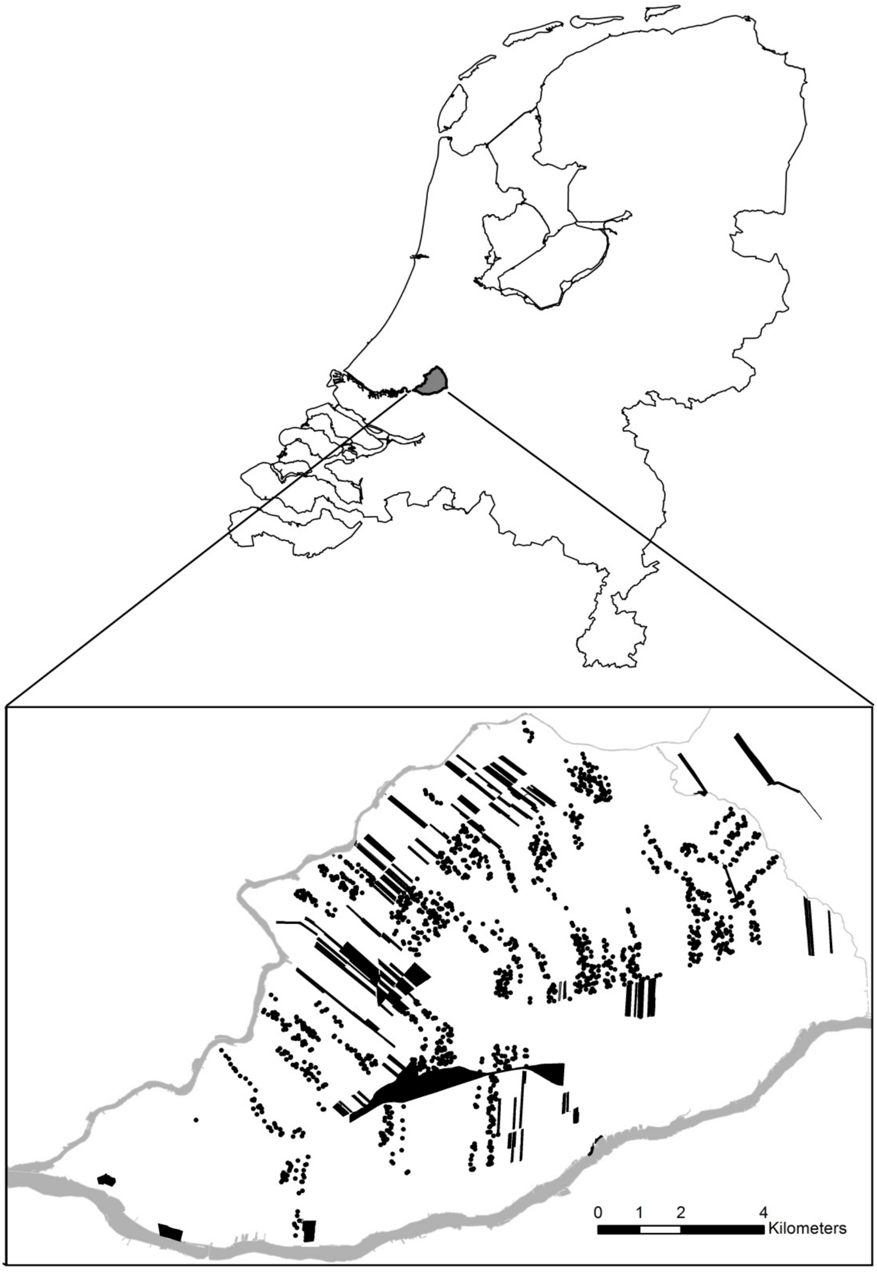

Figure 1. Location of the Krimpenerwaard in the Netherlands. Dots represent the 1494 quadrats in ditch banks with AES. Nature reserves in 1999 are shown with black polygons.

years (2000, 2005 and 2009), assuming that they were managed continuously in the intermediate years.

3.1. Nature reserve quadrats

The establishment of nature reserves in the Krimpenerwaard started in the early nineties of the twentieth century and increased from 674 hectares in 1999 to 1147 ha in 2005 and 1410 hectares in 2008. The majority of nature reserves consist of extensively managed grasslands for the conservation of meadow birds or grassland plant diversity. The type of management in the nature reserves depends upon its target group: for meadow bird conservation, reserves are fertilized with farmyard manure and extensively grazed, whereas management aimed at plant diversity consist of haymaking or grazing without fertilisation. We obtained data on the cover of plant species in quadrats in ditch banks of nature reserves from the foundation Zuid-Hollands Landschap, which manages these areas. We selected 122 permanent quadrats located in ditch banks of nature reserves in the study area, monitored between 1998 and 2009. This selection encompassed all quadrats that were monitored at least twice in the period 1998-2009 resulting in a total of 307 surveys. Quadrats in ditch banks of nature reserves were 50 m long and the width of the ditch bank was 0.49 m ± 0.15 m (average ± SD). To compare trends in nature reserves and AES ditch banks, we included only the target species that were also monitored in the AES ditch banks.

3.2. Calculating distance to nature reserves

We expect a decline in species richness with increasing distance from a nature reserve, due to an increasing seed limitation with distance. Previous research has shown that this decline with distance can best be approximated by an inverse power relationship between the distance to the seed source and seed abundance (Willson 1993, Coulson et al. 2001, Leng et al. 2009). In addition, we expect species to colonize adjacent ditch banks first

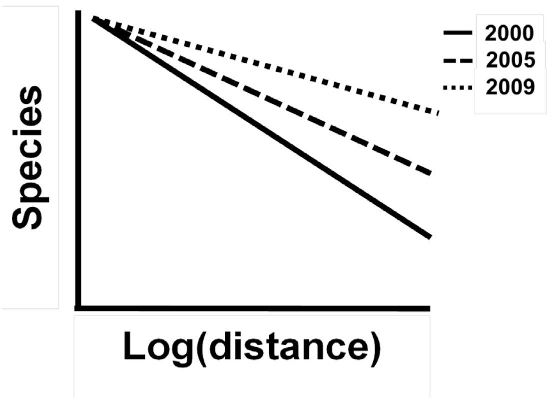

Figure 2. Hypothesized shifts over time in the relationship between plant species richness in ditch banks and the distance to a nature reserve after implementation of AES in 2000. Initially, increasing dispersal limitation with distance causes a strongly negative relationship between species richness and distance (solid line). With time, AES would allow plant species to establish (and set seed) in ditch banks at increasing distances from the reserve (dashed line, e.g. 2005), leading to an almost horizontal relationship in the long term (dotted line, e.g. 2009).

and start to spread seeds themselves at further distances, which will lead to a less steep distance-decay over time (Fig. 2).

We calculated the distance a seed had to travel from a nature reserve along ditches as follows. First, we imported a digital map of the ditches in the research area from the local water board into ArcGIS 10.1 (ESRI, Redlands, California; in 2011). In this map culverts (i.e. pipes that connect ditches when a road or passage for cattle runs across the ditch) were lacking. We added 6897 culverts from the national topography map of the

Netherlands of 2002 (Basis Registratie Topografie top10 vector map; (Kadaster 2002) to connect ditches. Based on aerial pictures (Eurosense B.V. 2008) we added another 22 culverts to link unconnected ditch banks to the network. Next we measured the distance from the quadrats in the ditch banks to the nearest nature reserves via this network.

Previous research has shown that wind is a more important determinant of the dispersal of floating seeds than the flow of water in slowly running waters like ditches (Soomers et al. 2010). Although the dominant wind direction in the research period was south-west in our research area, the variability in wind direction was high (SD = 91.7 degrees; N = 3653) (KNMI 2009). Therefore, we did not include dominant wind direction in our analyses. We calculated the distance from an AES quadrat to the nearest nature reserve that was created before 1999. We choose this methodology because recent nature reserves are often restoration projects that lack the high number of species typical of older nature reserves.

3.3. Plant dispersal categories

We divided target species into four different long distance dispersal strategies based on Hodgson et al. (1995). In this rating we distinguished only between capacity to disperse over long distances by means of water, wind or animals (Supplementary data Table A1). We considered Hodgson's categories "Species that shed their seeds from a capsule held above the vegetation that rocks in the wind" and "Species that disperse over short distances by ants" as having no-long distance dispersal capacity (Hodgson et al. 1995). The category of wind dispersal species comprised only one species (Cirsium palustre) and the category of animal dispersed species consisted of two species (Veronica beccabunga and Prunella vulgaris) that are rare in our research area (nomenclature following Van der Meijden (2005)) . Therefore we did not include these two dispersal categories in our analyses.

3.4. Statistics

As a first step, we analysed the number of target species present in ditch banks managed by farmers and those in nature reserves between 1998 and 2009. We performed a Generalized Linear Mixed model (GLMM) with a Poisson distribution and a log-link (Bolker et al. 2009). Management type (AES or reserve) was included as a fixed factor and year as a covariate. The interaction year x management was also included. To correct for the fact that data from the same nature reserve or farm may not be independent, nature reserve or farm was included as a random factor. We did not incorporate a repeated measures structure on the quadrat level, because AES quadrats were not permanent plots: they are redistributed yearly by the EC over a farmers ditch banks. We subsequently reduced the number of variables in the model based on their significance (type III sum of squares), until we obtained the simplest model with the lowest Akaike Information Criterion (AIC) value and significant terms only. This procedure was repeated for the number of target species within the two dispersal categories. As these analyses are based on a limited number of species (target species), we performed a similar analysis for all species in nature reserves to check for consistency. We repeated the GLMM as described above (but without management type) to analyse the total number of species and the number of species of each dispersal category.

Second, we tested the effect of distance to a nature reserve on the number of species in AES ditch banks, and how this effect changed with time, using the GLMM described above, but now distance to a nature reserve (log-transformed) and its interaction with year were also included. The total number of target species, and the numbers for each dispersal category, were analysed separately. Finally, we also analysed presence/absence of individual target species as a function of distance over time. This analysis was restricted to seven species that were present in more than 100 quadrats (out of almost 500) in 2000. We repeated the GLMM described above but with a binomial distribution and a logit-link function. All statistical analyses were performed using Predictive Analytics Software (PASW) (Version 19.0.03; SPSS/IBM Inc., Somers, NY, USA).

4. Results

4.1. Changes in species richness over time

The number of target species increased significantly in ditch banks under AES, but this

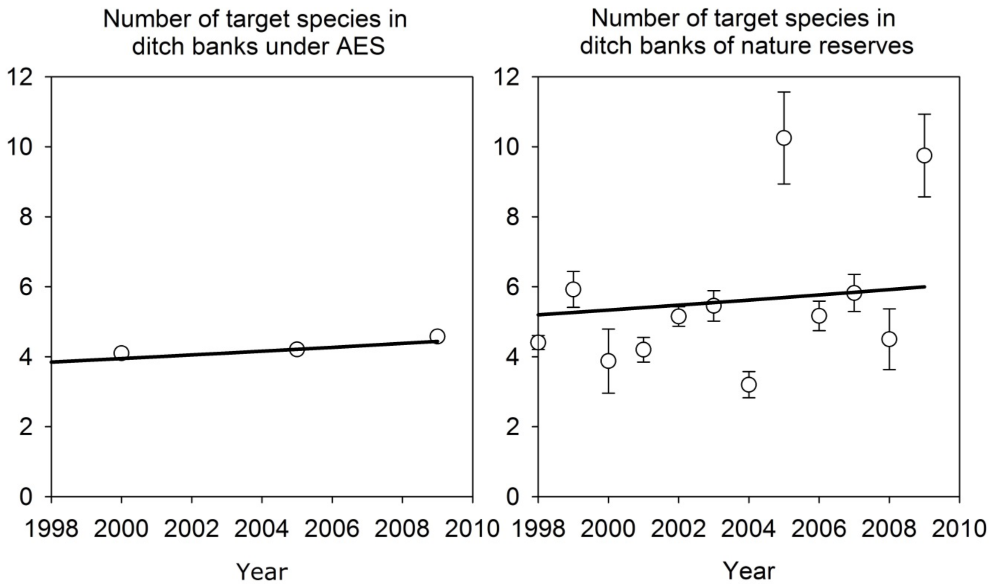

Figure 3. Changes over time in the number of target species in ditch banks under AES and ditch banks of nature reserves. Symbols represent the average per year with standard error. The line represent the predicted values over time determined by performing a GLMM analysis with management (AES or nature reserve), year and their interaction and subsequently removing factors until the simplest model was

Table 1. Results of the GLMM (F-values) testing the effects of management and year on tar get species richness and the number of target species able to disperse by water or with no capacity to disperse over long distances. Management was tested as a contrast AES ditch bank versus ditch bank in a nature reserve. Values are from the final model with significant terms only. NS indicate non-significant variables that were omitted from the model. Significance levels: *P<0.05, **P<0.01, ***P<0.001.

Group	df	Year	Management	Management*Year
Target species	1834	17.898***	17.650***	NS
Water LDD	1834	24.975***	15.863***	NS
No LDD	-	NS	NS	NS

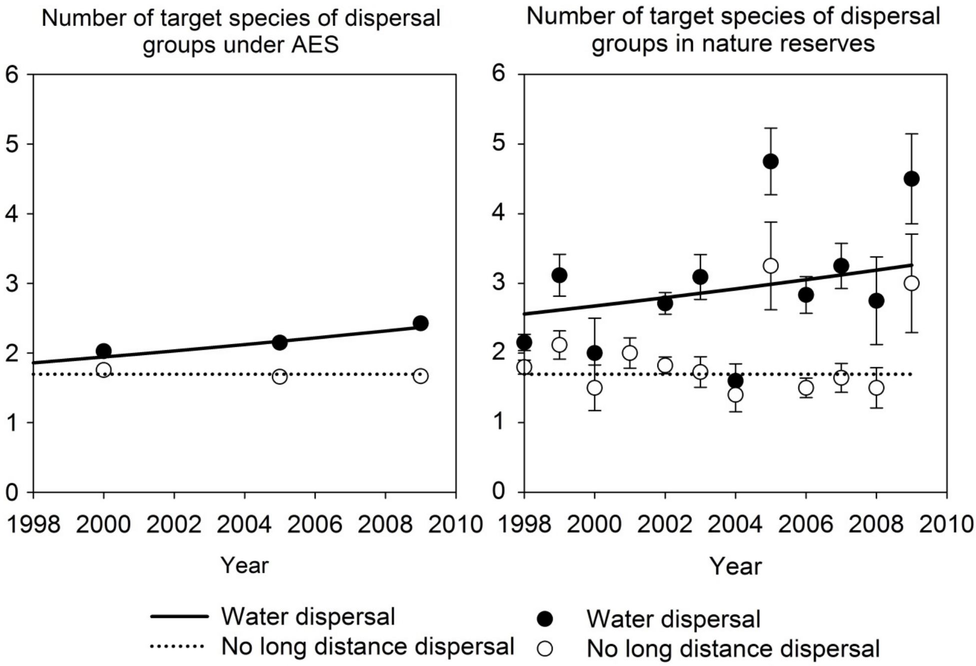

Figure 4. Changes over time of target species in ditch banks under AES and of ditches of nature reserves, divided into two groups according to dispersion strategy. Symbols represent the average number of target species per year with standard error. The line represent the predicted values over time determined by performing a GLMM analysis with management (AES or nature reserve), year and their interaction and subsequently removing factors until the simplest model was obtained with significant factors only. F-values and significance levels of the final model are summarized in Table 1. Notice that size of quadrats in nature reserves were 50m long and in AES ditch banks 100m.

also occurred in nature reserves (Table 1; Fig.3) The number of water-dispersed target species showed very similar results. They increased in ditch banks under AES and in nature reserves, but ditch banks of nature reserves contained more water-dispersed species (Fig.4). In contrast, species without long-distance dispersal showed no significant change in time in ditch banks under AES and in nature reserves while the number of species did not differ between ditch banks under AES and in nature reserves. As the target species are only a subset, we also analysed species richness of all plant species in nature reserves. In this case, the number of species did not increase with time (Supplementary data Figure A1), but we found a significant increase in time for the total number of water-dispersed species in nature reserves. The number of species without long-distance dispersal declined significantly ($F_{1,304}$:19.789 $P<0.001$) (Supplementary data Figure A2).

4.2. Effects of distance to a nature reserve

The average number of target species decreased significantly with an increasing distance from a nature reserve (Fig. 5, Table 2). As hypothesized, this decrease flattened out in

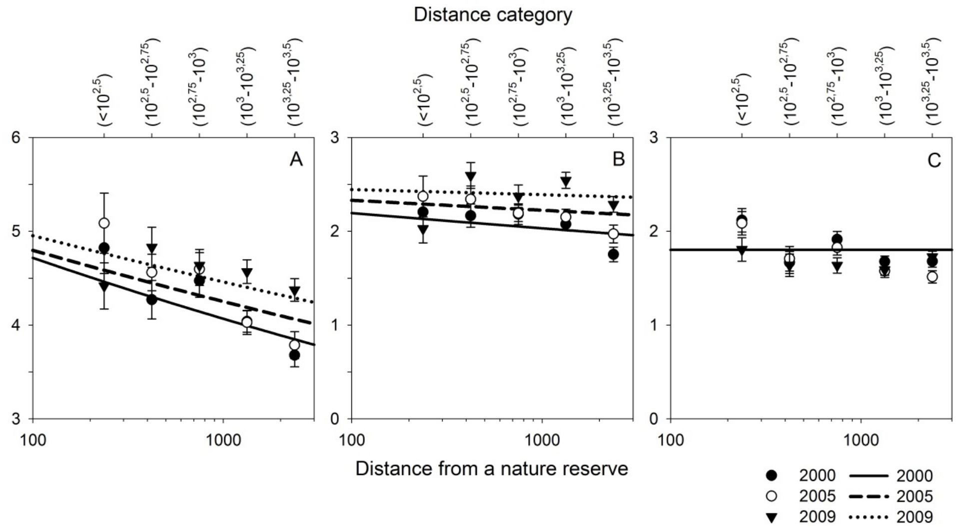

Figure 5. Changes in the number of species with distance from a nature reserve for all target species (A), water-dispersed species (B), and species with no long distance dispersal (C) in 2000, 2005 and 2009. Markers represent mean number of target species (± SE) per distance class. The distance classes (shown at the top) are chosen for illustration purposes only. The lines show the predicted values from the GLMM in which actual distances were used.

Table 2. Results of the GLMM (F-values) testing the effects of distance to a nature reserve and year on target species richness and the number of target species able to disperse by water or with no capacity to disperse over long distances. F-Values are obtained from the final model with significant terms only. NS indicate non-significant variables that were omitted from the model. Significance levels: *P<0.05, **P<0.01, ***P<0.001

Group	df	Year	Distance	Distance*Year
Target species	1491	NS	9.698**	14.33***
Water LDD	1491	NS	4.273*	19.585***
No LDD	-	NS	NS	NS

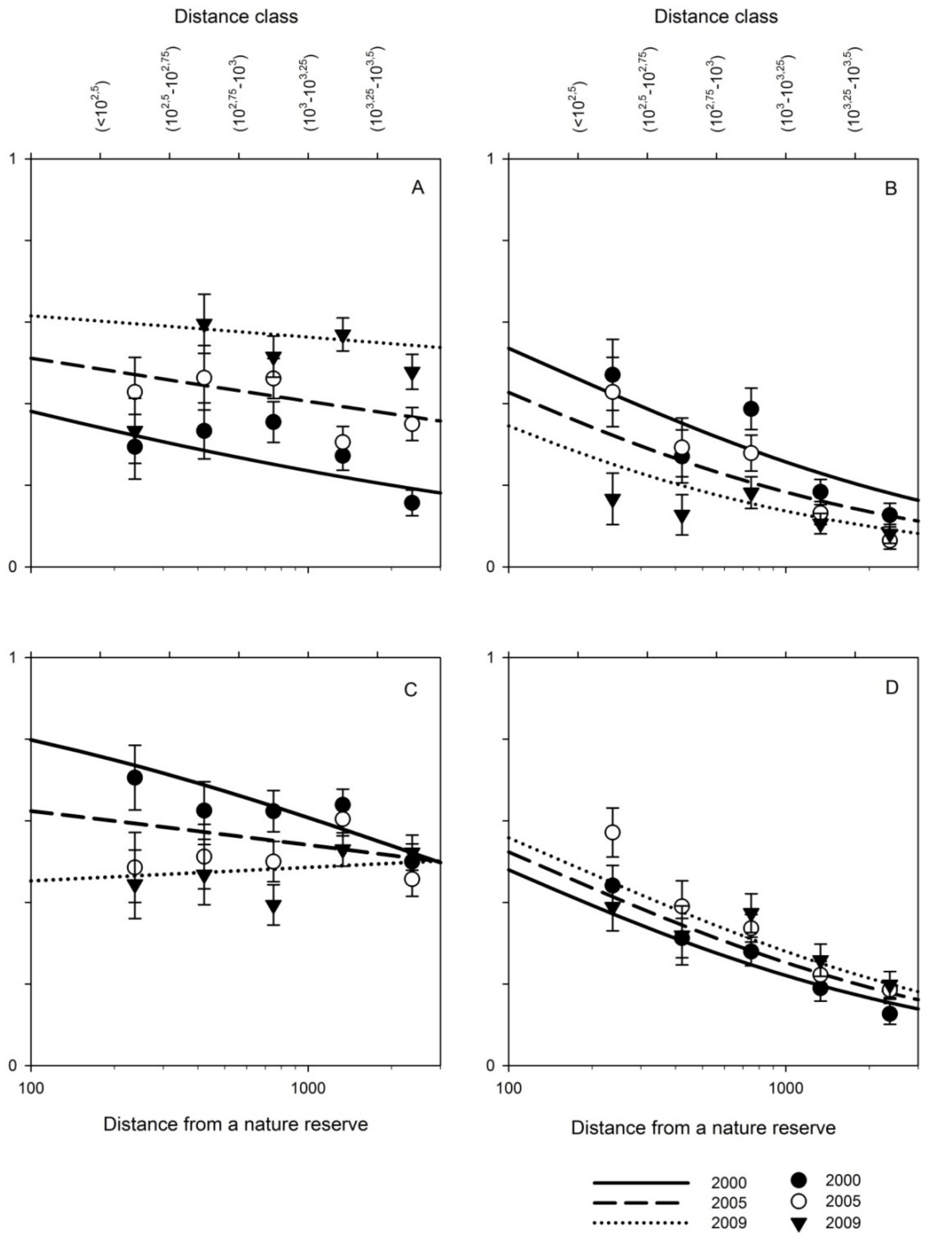

Figure 6. Changes in the presence of individual target species with distance from a nature reserve for Iris pseudacorus (A), Silene flos-cuculi (B), Ranunculus flammula (C) and Cirsium palustre (D) in 2000, 2005 and 2009. Markers represent mean number (± SE) per distance class. The distance classes (shown at the top) are chosen for illustration purposes only. The lines show the predicted values from the GLMM in which actual distances were used.

time demonstrated by the significant positive interaction between distance and year. This indicates the establishment of target species at longer distances from nature reserves. Separate analyses of the dispersal categories suggest that this pattern is due to water-dispersed target species, which constitute the largest group of species and also showed a decrease with distance that became less steep with time. We found no significant effects for species that do not have the capacity to disperse over long distances (Fig 5., Table 2).

Looking at individual species, Iris pseudacorus, which disperses by water, highlights the response of the water-dispersed category; a decrease with distance, but a strong increase in colonisation with distance over time (Fig 6a). The wind-dispersed Cirsium palustre showed a similar pattern, albeit less pronounced (Fig 6d). In contrast, typical hay meadow species declined over time. Silene flos-cuculi, which has no apparent adaptation to disperse over long distances, and the water-dispersed Ranunculus flammula showed the expected decrease with distance but declined strongly over time (Fig 6b,c). The other three species (Galium palustre, Lotus pedunculatus and Myosotis spp.) were very abundant (observed in 60 to 98% of the quadrants) and only showed small changes in time, but no effect of distance (Supplementary data Figure A3).

5. Discussion

The present study demonstrates that plant species richness has increased slightly in ditch banks under AES over the last ten years. These results corroborate previous research in this area, in which we found a small increase in plant species in agricultural ditch banks independent of AES (van Dijk et al. 2013a). Very similar patterns were observed in ditch banks in nature reserves, which makes it unlikely that the increase of target species in agricultural ditch banks is due solely to AES. In both studies, the increase in species richness could to a large extent be attributed to water-dispersed species, which confirms that dispersal capacity is an important functional trait in species-sorting in changing landscapes (Henle et al. 2004, Tscharntke et al. 2012).

Previous research has suggested that ditch banks can act as ecological corridors by which plant species can disperse from species-rich areas like nature reserves throughout the agricultural matrix (van Dorp et al. 1997, Geertsema et al. 2002, Donald and Evans 2006, Soomers et al. 2010). This study demonstrates a clear decline of plant species richness in ditch banks with increasing distance to nature reserves, which strongly suggests that nature reserves act as the main source for plant species richness in ditch banks under AES (Kohler et al. 2008, Leng et al. 2010b). In addition, our results show that the distance-decay became less steep with time, suggesting that during the last 10 year species established increasingly further away from nature reserves. This may be due to a cumulative effect of repeated colonization events from the same nature reserve, but could also reflect colonization from the individuals that first colonized the AES ditch banks. Detailed data on reproductive success of target species in AES ditch banks would be required to confirm this second explanation.

In line with earlier work (Soons et al. 2005, Kohler et al. 2007, Leng et al. 2010a, Soomers et al. 2010), these spatiotemporal patterns depended on the dispersal characteristics of the species. In our case, the pattern is mainly caused by water-dispersed species like I. pseudacorus. For other groups, we found no increase (species lacking adaptations to long-distance dispersal) or simply lacked sufficient species numbers (wind- and animal dispersed species). It is important to note that typical haymeadow species, like

S. flos-cuculi and R. flammula declined in the research period. This is unlikely to be driven by dispersal limitation alone (R. flammula is also dispersed by water), but probably reflects changes in environmental conditions in ditch banks. Earlier research has shown increased vegetation height and competition for light in these ditch banks (van Dijk et al. 2013b). This may indicate that for these species, site conditions have become less favourable under AES management.

Finally, it is important to point out that although we found significant increases in species richness, the effect size is very small (e.g. 0.5 target species in plots of 100 m x 1 m over 10 years in AES ditch banks at 3 km from a nature reserve). This demonstrates that although AES in ditch banks have the potential to act as corridors enhancing plant species richness, the dispersal process is either very slow or limited to a few species under current AES management. Recent studies indeed suggest that the increase in species numbers in ditch banks is limited to tall, fast-growing species instead of the species typical of the former species-rich hay-meadows that AES is often aimed at (van Dijk et al. 2013a, van Dijk et al. 2013b). The efficacy of AES may be enhanced by measures aiming at decreasing nutrient availability and competition for light, e.g. by increasing the rate of hay-making. But also measures that enhance dispersal should be promoted (Leng et al. 2011). Finally, increasing the width of the ditch bank may increase the colonization success in AES ditch banks (van Dorp et al. 1997, Soons et al. 2005).

Acknowledgements
We would like to thank Rianne de Wit of EC Weidehof Krimpenerwaard for providing data of AES quadrats and Dick Kerkhof of Zuid Hollands Landschap for providing data on nature reserves. Also, we would like to thank Jan Clement for helping with the spatial analyses. This research is part of the KBIV strategic research programme “Sustainable spatial development of ecosystems, landscapes, seas and regions” which is funded by the Dutch Ministry of Economic Affairs, Agriculture and Innovation, and carried out by Wageningen University & Research centre.

Supplementary Material

Table A1. List of target species monitored in Weidehof Krimpenerwaard. Columns Wind, Water, Animal and No Long Distance Dispersal (LDD) show a species' capacity to use the mentioned vector for long distance dispersal based on (Hodgson et al. 1995). Dashes indicate no data on long-distance dispersal vector available. Columns 2000, 2005 and 2009 shows the number of quadrats in which a species was present.

	Wind LDD	Water LDD	Animal LDD	No LDD	2000	2005	2009
Achillea ptarmica	0	0	0	1	0	1	1
Caltha palustris	0	1	0	0	14	12	13
Centaurea jacea	-	-	-	-	0	0	0
Cirsium dissectum	-	-	-	-	1	0	0
Cirsium palustre	1	0	0	0	107	135	134
Filipendula ulmaria	0	1	0	0	8	10	14
Galium palustre	0	1	0	0	457	475	487
Hydrocotyle vulgaris	0	1	0	0	7	10	7
Iris pseudacorus	0	1	0	0	133	192	264
Lathyrus pratensis	0	0	0	1	4	2	7
Leucanthemum vulgare	0	0	0	1	0	0	0
Lotus pedunculatus	0	0	0	1	301	328	337
Lysimachia thyrsiflora	-	-	-	-	24	36	79
Lysimachia vulgaris	0	1	0	0	3	2	2
Lythrum salicaria	0	1	0	0	22	40	33
Mentha spp.[a]	0	1	0	0	74	68	150
Myosotis spp.[b]	0	0	0	1	453	407	423
Pedicularis palustris	-	-	-	-	0	0	0
Potentilla palustris	0	0	0	1	0	0	0
Prunella vulgaris	0	0	1	0	3	2	0
Ranunculus flammula	0	1	0	0	290	260	238
Rhinanthus angustifolius	-	-	-	-	0	0	0
Silene flos-cuculi	0	0	0	1	116	88	61
Veronica beccabunga	0	0	1	0	14	11	20
Vicia cracca	0	0	0	1	12	18	10
Total	1	9	2	8	2043	2097	2280

Lumped taxa [a] Mentha aquatica and Mentha arvensis, [b] Myosotis scorpioides subsp. scorpioides and Myosotis laxa subsp. cespitosa.

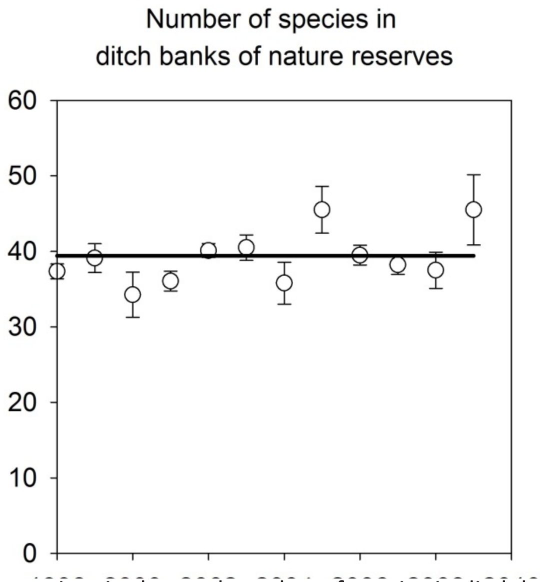

Figure A1. Changes over time in the total number of species in ditch banks of nature reserves. Symbols represents the average per year with standard error. The line represent the predicted values over time determined by performing a GLMM analysis with year and subsequently removing factors until the simplest model was obtained with significant factors only.

Table A2. Summary of Generalized Linear Mixed models (F-values) testing the effects of distance to a nature reserve and year on individual target species that were present in at least 100 quadrats in 2000. F-Values are obtained from the reduced model with significant terms only. NS indicate non-significant variables that were omitted from the final model.

Species	df	Year	Distance	Distance*Year
Iris pseudacorus	1491	NS	8.339**	89.280***
Galium palustre	1492	18.737***	NS	NS
Lotus pedunculatus	1492	7.301**	NS	NS
Myosotis spp.	1492	9.079**	NS	NS
Ranunculus flammula	1490	4.529*	9.624**	12.346***
Silene flos-cuculi	1491	24.267***	11.338**	NS
Cirsium palustre	1491	5.292*	14.210***	NS

Significance levels: *P<0.05, **P<0.01, ***P<0.001

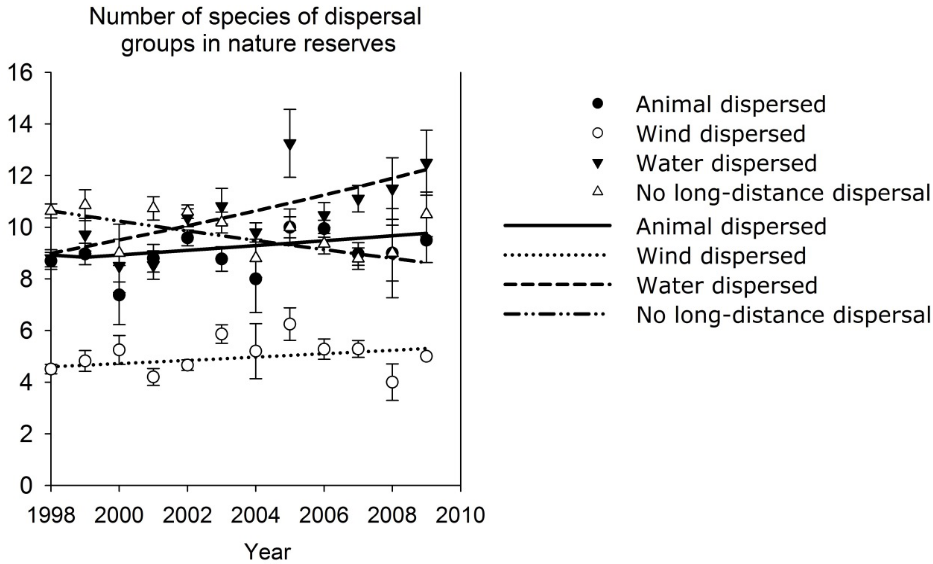

Figure A2. Changes over time in dispersal categories in ditch banks based on relative abundance of all species and species richness in ditch banks of nature reserves. Symbols represents the average per year with standard error. The line represent the predicted values over time determined by performing a GLMM analysis with year as covariate and subsequently removing factors until the simplest model was obtained with significant factors only for abundance and species richness.

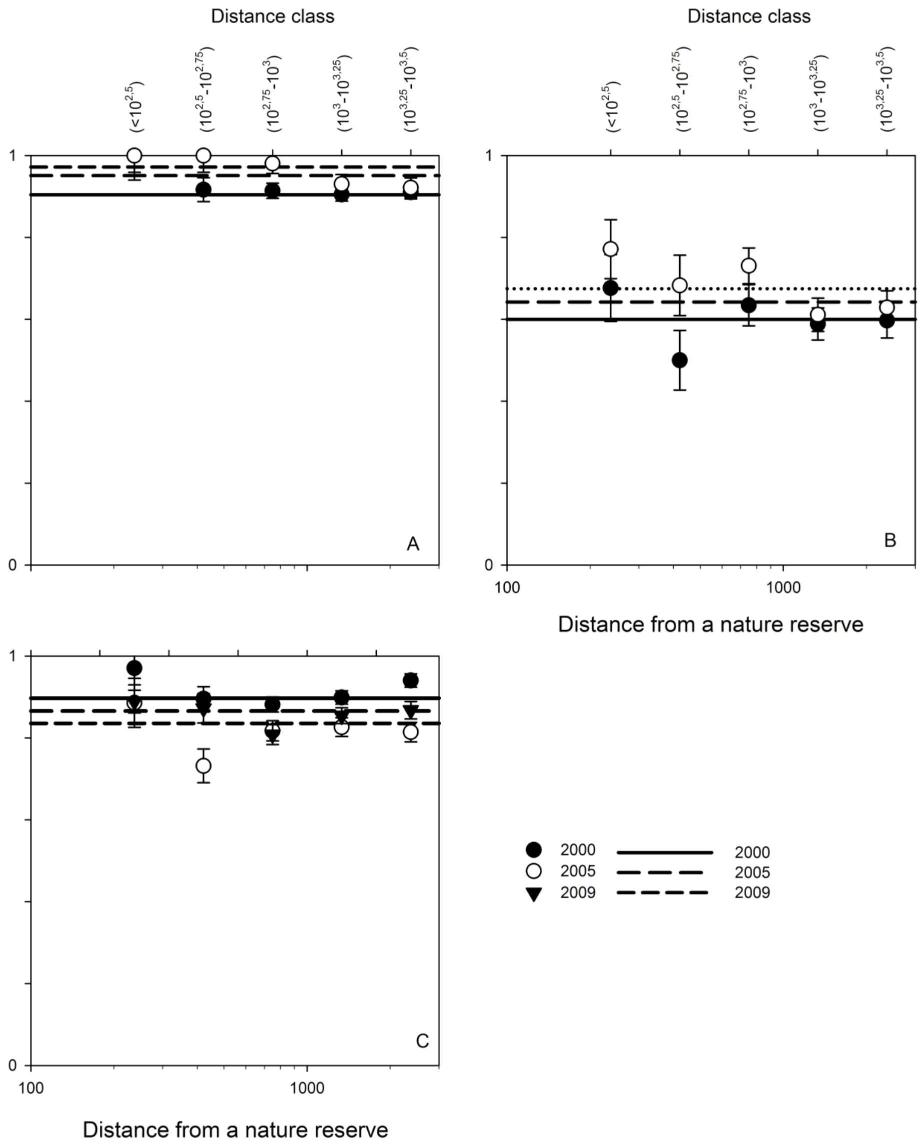

Figure A3. Changes in the presence of individual target species with distance from a nature reserve for Galium palustre (A), Lotus pedunculatus (B) and Myosotis spp. (C) in 2000, 2005 and 2009. Markers represent mean number (± SE) per distance class. The distance classes (shown at the top) are chosen for illustration purposes only. The lines show the predicted values from the GLMM in which actual distances were used. Mentha spp. is Mentha aquatica and Mentha arvensis lumped together. Myosotis spp. is Myosotis scorpioides subsp.scorpioides and Myosotis laxa subsp. cespitosa lumped together.

4. Collective agri-environment schemes: How can membership of regional environmental cooperatives enhance farmers’ intentions for agri-environment schemes?

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In review in Land Use Policy

Abstract
The effectiveness of Agri-Environment Schemes (AES) in enhancing biodiversity on farmland and creating a long-lasting change in farmers' motivation towards a more environmental-friendly practice is still strongly debated. Applying a regional approach has been advocated widely to make AES more ecologically and socially sustainable.
In the Netherlands, some AES are performed collectively by large regional groups of farmers called Environmental Cooperatives (EC). It has been hypothesized that these cooperatives enhance farmers' motivation to perform AES by facilitating the application of AES, but also by generating group pressure. In the study at hand, we used an extended version of the Theory of Planned Behaviour (TPB) to investigate which factors are associated with farmers' intention to perform two kinds of collective AES (ditch bank management and the protection of meadow birds). Our results demonstrate that attitude and perceived personal ability to perform these AES are associated with the intention of farmers to perform ditch bank management. However, for the protection of meadow birds, social pressure, self-identity and facilitation by the EC also relate to the intention of farmers. We conclude that there is an advantage of ECs for the performance of collective AES by means of facilitation.

1. Introduction

Biodiversity is declining worldwide at a vast rate due to the loss of natural areas (Millennium Ecosystem Assessment 2005). Europe, after having failed to halt biodiversity loss by 2010, has set the goal to halt biodiversity loss by 2020 (EEA 2006, Maes et al. 2012). One of the main drivers of biodiversity loss in Europe is the intensification of agricultural practice over the last decades (Stoate et al. 2001, Stoate et al. 2009). Farmland covers about 45% of rural areas in Europe and many threatened species are associated with agricultural habitats (Kleijn et al. 2011). In order to realize enough area to stop biodiversity decline, integration of agriculture within EU nature conservation policy is inevitable (Siebert et al. 2006, Balmford et al. 2012). This integration has been sculpted by implementing agri-environment schemes (AES) on European farmland.

In Europe €5 billion is annually spent on AES (Kleijn et al. 2011, Balmford et al. 2012). This amount is similar to the costs of the European-wide Natura 2000 Network of nature reserves (Gantioler et al. 2010), despite the common notion that AES are cheaper (Jongeneel et al. 2012). However, the results of AES in conserving biodiversity vary widely (Kleijn et al. 2001, Kleijn and Sutherland 2003, Kleijn et al. 2011).

1.1. Environmental cooperatives

One of the major points of criticism to current AES is that a landscape approach is required in order to make AES effective (Kleijn et al. 2011, Tscharntke et al. 2012, van Dijk et al. 2013a). In order to perform AES in a landscape context two factors are crucial; farmers' willingness to cooperate and a landscape-wide organisational structure to coordinate the implementation of AES (Beedell and Rehman 2000). In current policies, expected cooperation of farmers is often based on a solely economic rationale. This may be too simplistic; farmers make decisions in an economic, but also in a social and cultural context (Burton 2004, Sutherland 2010). In the Netherlands, where some forms of AES started as early as 1980 (Berendse et al. 2004), the first farmer cooperatives coordinating agri-environmental measures were established in 1991 (Oerlemans et al. 2006). These so-called Environmental Cooperatives (EC) originated from the growing concern amongst farmers about the direction the Dutch agri-environment programme was heading. Their main points of criticism were the exclusion of farmers' opinions, a top-down approach and farmers' alleged lack of responsibility for the environment (Groeneveld et al. 2004, Franks and Mc Gloin 2007). From 2000 on, the Dutch Agri-environmental program (Programma Beheer) created possibilities for ECs to apply for regional agri-environmental schemes. These AES often cover more than 100 hectares and involve multiple farmers (Oerlemans et al. 2007). In 2006 these collective AES made up 39% of the total area of AES and covered 34% of the total Dutch expenditure on AES (Oerlemans et al. 2006). This strengthened the mediating role that the ECs played between the government and individual farmers in the application for collective AES (Glasbergen 2000).

ECs can benefit the government in various ways: They lower administration costs, and they form a single contact point, essential for advice and representation of their members (Franks and Mc Gloin 2007). Previous research demonstrated that individual farmers labelled these ECs as the most influential stakeholder involved in AES, while for instance nature conservation agencies were graded lowest (Noordijk et al. 2009).

Due to these benefits the concept of Dutch farmers cooperatives has been proposed in the UK (Franks and Mc Gloin 2007, Mills et al. 2011, Emery and Franks 2012, Mills 2012), Europe-wide (Burton and Schwarz 2013) and in other continents (Attwood et al. 2009, McKenzie et al. 2013). Current reforms of the European Common Agricultural Policy even include the introduction of AES at a regional level, after successful pilots in the Netherlands with ECs (European Commission 2011, IEEP 2012).

The advantages of ECs for their members have been discussed in a large body of literature (de Snoo et al. 2010, Mills et al. 2011, de Snoo et al. 2012a, Mills 2012, Franks and Emery 2013) for an overview see (Franks and Mc Gloin 2007). Such advantages include the support of a group of environmentally minded farmers to resist pressure of other farmers that are more production minded (Burton and Paragahawewa 2011, Mills et al. 2011), but also more practically, help with application for participation in the schemes (Franks and Mc Gloin 2007).

However, most of these studies are from a qualitative point of view, which makes comparative analyses difficult. Moreover, while these studies shed light on possible advantages of EC membership as experienced by farmers, they do not test whether these advantages actually lead to changes in motivation. So far, no quantitative empirical study on the relation between membership of an EC and individual farmers' intentions to perform AES has been performed. The goal of the current paper is to study how being member of an EC affect individual farmers' intentions to perform agri-environmental measures. To do so we will use an adapted and extended version of the Theory of Planned Behaviour (TPB;(Ajzen 1991)) as a framework.

1.2. Our model

The Theory of Planned Behaviour states that the intention to perform a certain behaviour is determined by three factors: attitudes one has towards this particular behaviour, subjective norms (the perceived social pressure that one feels from significant other people to perform this behaviour) and perceived behavioural control (the perceived ability that one feels to perform this behaviour). These three variables are driven by evaluations and beliefs about the results of the behaviour (attitude), the groups and persons who are regarded as significant others (subjective norm) and the skills and barriers one thinks are supporting or opposing the performance of the behaviour (perceived behavioural control).

The TPB has been demonstrated to provide a structured yet flexible model that can explain the cognitions that underlie individual farmers' willingness to perform AES (Burton 2004, Fielding et al. 2005, Sutherland 2010, Wauters et al. 2010, Lokhorst et al. 2011). The TPB is flexible because it is "in principle, open to the inclusion of additional predictors if it can be shown that they capture a significant proportion of the variance in intention or behaviour after the theory's current variables have been taken into account." (Ajzen 1991).

One of the most prominent additions to TPB is the inclusion of self-identity as a predictor of intention (Conner and Armitage 1998, Terry et al. 1999). The effect of self-identity on intention derives from identity-theory (Stryker 1968). This theory states that the self consists of various identities based on the social role that one occupies. In different situations, different identities may be most salient to affect behaviour. In TPB self-identity is defined as the extent to which a certain behaviour is considered being part of the self (Terry and Hogg 1996, Terry et al. 1999). Self-identity has been demonstrated to play a significant role in farmers' intention to perform AES (Fielding et al. 2008, Lokhorst et al. 2011, Mastrangelo et al. 2013), such that the more farmers see conservation as part of the self, the more likely they are to intend to engage in AES.

Previous research showed that also a number of predictors related to membership of a group can have an effect on one's intention to perform a certain behaviour (Terry and Hogg 1996, Terry et al. 1999). In the study at hand, we will investigate how the membership of an EC explains variation of intention to perform AES and how this is incorporated in TPB.

The most prominent addition to measure the influence of relevant groups for behaviour is the inclusion of constructs from social-identity theory, namely group norms and group identification (Terry et al. 1999). Previous research has demonstrated that of the standard TPB variables, subjective norms are generally the weakest predictors of intentions (for an overview see (Armitage and Conner 1999). The conceptualization of norms has been hypothesized as being the reason of this; Rather than the total pressure we perceive from important others, the pressure we feel from peer groups relevant to the behaviour would influence the intention to perform that behaviour (Terry et al. 1999). However, this pressure is only perceived if one identifies strongly with the particular peer group (Terry and Hogg 1996, Terry et al. 1999). Group norms have been demonstrated to affect the intention of farmers in collectives to perform a more sustainable agricultural practice, but this effect depended on the level to which farmers identify with the group (Beedell and Rehman 2000, Franks and McGloin 2007, Fielding et al. 2008). So, it seems reasonable to assume that in the case of performing AES, EC's are seen as the most relevant peer group. After all, they consist of other local farmers engaged in AES, and decisions concerning AES are often made with the EC. This may lead to members experiencing group pressure to perform AES (Beedell and Rehman 2000, Fielding et al. 2005, Franks and McGloin 2007, Fielding et al. 2008, Burton and Schwarz 2013).

Besides group pressure, ECs have also been hypothesized to facilitate the performance of AES by their members (Franks and McGloin 2007, Emery and Franks 2012), and this may then lead to an increase in individuals perceived behavioural control. However, several investigators have questioned the reliability of the traditional perceived behavioural items in covering both personal ability to perform a certain behaviour and the facilitation of external factors in performing this behaviour (Conner and Armitage 1998, Manstead and van Eekelen 1998, Ajzen 2002). In the case of ECs, group facilitation would not be a personal characteristic anymore, and thus should be measured separately.

In the present study we will add three variables into TPB to see how membership of an EC is associated with farmers' intentions to perform AES. First, we expect the traditional TPB constructs attitude and perceived behavioural control to be positively associated with intention to perform AES (H1). Second, we expect farmers who relate AES to their self-identity to have a higher intention to perform AES (H2).

Third, we expect group norms, rather than subjective norm to be associated with a higher intention (H3), especially for farmers that identify strongly with the EC (H4). Next, we expect farmers that perceive a high group facilitation by the EC to have a higher intention to perform AES (H5).

Previous research has demonstrated that farmers interests, knowledge and willingness to perform AES deviates between AES focussing at meadow birds and at plants (Herzon and Mikk 2007). Therefore, we will measure all variables in relation to 2 kinds of nature management; conservation of meadow birds and conservation of plants in ditch banks.

2. Material and methods

2.1. Procedure and Participants

We tested a first version of the questionnaire on 5 different farmers to ensure a comprehensible phrasing of the questions. We then sent 541 surveys by mail to members of 4 ECs in the Western part of the Netherlands. We obtained the postal addresses of farmers participating in AES from the ECs. We added a meadow bird identification poster with the questionnaires to increase the response rate. After 2 weeks we sent a reminder email. After 2 months we called participants who had not returned the questionnaire to ask them to complete the survey after we had resent it. A total of 297 surveys were returned, representing a 55% response rate. Out of these a total of 185 farmers performed collective ditch bank management between 2000 and 2009 and 247 farmers performed collective meadow bird management.

2.2. Questionnaire

All items on ditch bank management in our questionnaire were focused on subsidized ditch bank management between 2000 and 2009, because then the ECs still facilitated this management by means of collective contracts. For meadow bird management we inquired into current management, which is still performed collectively. Because we measured the TPB items for two kinds of AES and still wanted to obtain a response rate as high as possible we used modified scales with fewer items. All items were measured on 5-point Likert scales. We used 2 items to measure intention (Perugini and Bagozzi 2001): "In the future, under the same conditions I would perform ditch bank management / meadow bird management again" and "Given the opportunity I would perform ditch bank management / meadow bird management again: certainly not / certainly". This scale yielded a sufficient high reliability for both the intention to perform ditch bank management (Cronbach's α=0.74) and meadow bird management (Cronbach's α=0.75). We measured attitude for both forms of agri-environmental measures using the items "I think that ditch bank management / meadow bird management is: negative / positive", "I think that ditch bank management/ meadow bird management is: useless / useful", and "I think that ditch bank management / meadow bird management is: unimportant / important" (Ajzen and Fishbein 1980). This scale yielded only a sufficiently high reliability for ditch bank management (Cronbach's α=.85). For meadow bird management, even when deleting one item we did not obtain sufficiently high reliability for this scale (α = 0.62). Therefore, for meadow bird management we used a single item operationalization with "I think that meadow bird management is: useless / useful" in our analysis. For subjective norm we used the item "Most people who are important to me think it is important that I carry out ditch bank management / meadow bird management: completely disagree / completely agree" (Ajzen and Fishbein 1980).

To measure perceived behavioural control (PBC) we used the following item "I am capable of carrying out ditch bank management / meadow bird management: completely disagree / completely agree" (Ajzen and Fishbein 1980).

We used 2 items to measure self-identity: "ditch bank management / meadow bird management is part of who I am" and "ditch bank management / meadow bird management is something that is typical for me" (Terry et al. 1999, Lokhorst et al. 2011). For these scales we obtained Crohnbach's α of 0.83 for both ditch bank management and meadow bird management.

To measure group norms of the EC we specified the measure of group norm to EC as reference group by using the item "Most members of my EC think it is important that I carry out ditch bank management / meadow bird management: completely disagree / completely agree".

To measure the facilitation of the EC we used the item " Thanks to the EC, I am capable of carrying out ditch bank management / meadow bird management: completely disagree / completely agree".

To measure identification with the EC we used 4 items (Mael and Ashforth 1992); "I am affected if members of my EC are being criticized", "If members of my EC are thriving I am happy", "I am proud of the members of my EC" and "When I'm talking about the members of my EC, I'm talking about "we" and not "them". For these scales we obtained Crohnbach's α of 0.77 for farmers that perform ditch bank management and 0.81 for farmers that perform meadow bird management.

2.3. Statistics

We performed a hierarchical regression with intention to perform ditch bank management or meadow bird management as the dependent variable. In the first step, we entered the demographic variables age, level of education, farm size and farmers' perception of financial profitability of the AES performed. Next, we included the traditional TPB variables attitude, subjective norm and PBC. In the third step we added the variable self-identity. Fourth, we included group norm and group identity, and in the subsequent step their interaction. Next, we added group facilitation and finally the interaction of group facilitation with group identity. To avoid distortion of the results caused by multicollinearity between the predictors and the interaction terms, we based the interaction terms on mean-centred values of the variables (Aiken and West 1991, Terry et al. 1999).

3. Results

For both ditch bank management (Table 1) and meadow bird management (Table 2) we found significant positive correlations among all the measured variables with intention to perform that particular AES. However, for meadow bird management all these correlations had higher r-values than for ditch bank management. Particularly, our additions to the

Table 1 Means, standard deviations and correlations of variables in our model for ditch bank management. SN represents subjective norm and PBC perceiver behavioural control.

		M	SD	1	2	3	4	5	6	7	8
1	Intention	4.29	0.95	1	$.50^{**}$	$.24^{**}$	$.41^{**}$	$.32^{**}$	$.18^{*}$	$.22^{**}$	$.15^{*}$
2	Attitude	4.22	0.86		1	$.28^{**}$	$.52^{**}$	$.52^{**}$	$.26^{**}$	$.39^{**}$	$.25^{**}$
3	SN	3.23	1.18			1	$.34^{**}$	$.50^{**}$	$.46^{**}$	$.41^{**}$	$.27^{**}$
4	PBC	4.42	0.87				1	$.59^{**}$	$.40^{**}$	$.44^{**}$	$.26^{**}$
5	Self-Identity	3.69	1.04					1	$.49^{**}$	$.42^{**}$	$.39^{**}$
6	Group norm	3.63	1.08						1	$.41^{**}$	$.31^{**}$
7	Group facilitation	4.03	0.99							1	$.34^{**}$
8	Group identification	3.77	0.78								1

Table 2 Means, standard deviations and correlations of variables in our model for meadow bird management. SN represents subjective norm and PBC perceived behavioural control.

		M	SD	1	2	3	4	5	6	7	8
1	Intention	4.16	0.84	1	.66**	.44**	.65**	.66**	.37**	.50**	.28**
2	Attitude	4.28	0.93		1	.30**	.37**	.56**	.38**	.29**	.15*
3	SN	3.58	1.00			1	.22**	.45**	.53**	.32**	.48**
4	PBC	3.90	1.08				1	.58**	.28**	.48**	.26**
5	Self-Identity	3.65	0.88					1	.47**	.45**	.40**
6	Group norm	3.74	0.99						1	.53**	.47**
7	Group facilitation	4.05	1.01							1	.37**
8	Group identification	3.72	0.75								1

standard TPB, self-identity, group norm, group facilitation and group identity, demonstrate much higher correlations with intention for meadow bird management, than for ditch bank management.

3.1. Ditch bank management

We performed a hierarchical regression with intention to perform ditch bank management as the dependent variable. None of the four demographic variables explained a significant part of the variation in the intention to perform ditch bank

Table 3 Hierarchical regression analyses predicting intention to perform ditch bank management. GIxGn is the interaction group identification group norm. Significance levels: *P<0.05, **P<0.01, ***P<0.001

	R^2	ΔR^2	1	2	3	4	5	6
Age	.025		-.011	-.008	-.008	-.009	-.008	-.008
Education			.033	.019	.017	.013	.012	.014
Farmsize			.003	.004	.004	.004	.004	.004
Profitability			.081	.017	.014	.011	.001	.003
Attitude	.294	.269***		.433***	.451***	.450***	.443***	.448***
Subjective norm				.032	.045	.057	.058	.063
PBC				.206*	.231*	.241*	.245*	.252*
Self-identity	.296	.002			-.057	-.041	-.036	-.035
Group norm	.298	.002				-.046	-.042	-.035
Group identification						-.017	-.023	-.017
GIx GN	.300	.002					.046	.048
Group facilitation	.301	.001						-.034

management. In the second step, attitude (B=0.432, p<0.001) and perceived behavioural control (B=0.267, p<0.01) were both significant predictors of the intention to perform ditch bank management, but subjective norm was not (Table 2).

The TPB variables together explained 29.8% of the variation in intention to perform AES. To test for the variables derived from identity theory we first added self-identity, then group-identification and group-norm, their interaction and finally group facilitation. None of these factors were significantly associated with intention, neither did these steps explain any significant additional variation.

3.2. Meadow bird management

We repeated the procedure of ditch bank management for meadow bird management. No significant association between age, level of education, farm size or perceived profitability of AES and intention to perform meadow bird management was obtained (Table 4). In the second step, attitude (B=0.394, p<0.001), subjective norm (B=0.179, p<0.001) and perceived behavioural control (B=0.179, p<0.001) showed a significant association with the intention to perform meadow bird management. This step explained 66.5% of the variation in intention to perform meadow bird management.

Table 4 Hierarchical regression analyses predicting intention to perform meadow bird management. GIxGn is the interaction group identification group norm. Significance levels: *P<0.05, **P<0.01, ***P<0.001

	R^2	ΔR^2	1	2	3	4	5	6
Age	.041		-.014*	-.001	.000	.001	.001	.001
Education			.008	.023	.019	.021	.021	.027
Farmsize			.003	.000	.000	.000	.000	.000
Profitability			.066	-.021	-.024	-.025	-.026	-.027
Attitude	.665	.624***		.394***	.348***	.358***	.358***	.363***
Subjective norm				.179***	.147***	.180***	.180***	.186***
PBC				.342***	.293***	.295***	.295***	.251***
Self-identity	.677	.012*			.150*	.170**	.170**	.159*
Group norm	.682	.005*				-.068	-.068	-.125*
Group identification						-.018	-.017	-.033
GIx GN	.682	.000					.006	.011
Group facilitation	.699	.017**						.143**

Subsequently, we added self-identity to the model, which was significantly associated with intention (B=0.150, p<0.05) and this step explained significantly more variation in intention (1,2%, $F_{1,162}$=5.821, P<0.05). Next the social identity theory variables were added to the model. Group norm, group-identification and their interaction had no significant relation with intention, nor explained any additional variation. Finally, group facilitation was added, which had a significant positive association with intention (B=0.143, p<0.01) and explained an additional 1,7% of variation in intention ($F_{1,158}$=9.142, P<0.01). In this step group norm changed from no significant association with intention to a significant negative association with intention (B=-0.125, p<0.05). Contrary to our expectations, in the final step group norm changed from no significant association with intention to a significant negative association with intention (B=-0.125, p<0.05), while in the bivariate analysis, group norm and intention were positively correlated with each other (r=0.37, p<0.001, see Table 3). The strong correlation between group norm and group facilitation (r=0.53, p<0.001) indicates a confounding effect. To test for this we repeated our model without group norm and the interaction group norm, group identity. In this model, group facilitation had a significant positive association with intention (B=0.092, p<0.05) and adding group facilitation explained an additional 1% of the variation in intention ($F_{1,160}$=5.167, P<0.05). This indicates that, even without group norm, group facilitation has a significant direct association with intention. Group norm, on the other hand, had no significant association with intention if group facilitation is not included in the model. Only if group facilitation was included, the confounding effect between group norm and group facilitation resulted in a significant association of group norm with intention

3.3. Hypothesis testing

There was a large difference in the explained variation in intention for ditch bank management (29%) and meadow bird management (69%). Our results demonstrate that attitude and perceived behavioural control explain a significant amount of the variation of farmers intention to perform both ditch bank management and meadow bird management, confirming hypothesis 1. Self-identity had a significant relation with farmers' intention to perform meadow bird management, but not on the intention for ditch bank management, partially supporting hypothesis 2.

Contrary to hypothesis 3, group norm did not replace subjective norm as determinant of intention. On the contrary, subjective norms significantly enhanced farmers' intention to perform meadow bird management, while group norm did not. Neither did the relation between group norm and intention differ between farmers that identify strongly or weakly with the EC, rejecting hypothesis 4. We did find a significant association between group facilitation and intention to perform meadow bird management, confirming hypothesis 5. For ditch bank management we found no significant relation between either group identification, group norm, their interaction or group facilitation with intention.

4. Discussion

Adopting a landscape approach by means of collective AES have been proposed by various authors as a possible improvement that would make AES more ecologically and socially sustainable (Burton and Paragahawewa 2011, Kleijn et al. 2011, de Snoo et al. 2012a, Mills 2012, Tscharntke et al. 2012, van Dijk et al. 2013a). Regional socially embedded organisations such as the Dutch environmental cooperatives have been advocated in other countries as a facilitator of collective AES (Franks and Mc Gloin 2007, Burton and Paragahawewa 2011, Mills 2012). Here, we demonstrate how membership of such EC's affects individual farmers' intentions to perform subsidized AES. To the best of our knowledge, this is the first attempt to tackle this important issue empirically.

Our findings confirm the hypothesised facilitation of the EC of performing AES

(Franks and Mc Gloin 2007, Franks and McGloin 2007, Mills et al. 2011, Franks and Emery 2013). Previous research has shown that farmers perceive this facilitation as for example the exchange of information and knowledge within the collective (Franks and Mc Gloin 2007, Mills et al. 2011). This has been theorized to open opportunities for the creation of both cultural (knowledge, skills and ability) and social (access to social networks and their resources) capital, resulting in symbolic capital (pride and prestige) among the farming community for AES (Burton and Wilson 2006, Burton and Paragahawewa 2011, Sutherland and Burton 2011). This would make collective AES more social and cultural sustainable (Burton et al. 2008, Burton and Paragahawewa 2011, Burton and Schwarz 2013). The current study adds to this debate by explicitly showing that the facilitation of the EC is directly associated with farmers' intention to perform collective AES.

We found no relation between group pressure of the EC and their members' intention to perform meadow bird management, while general subjective norms did play a role. This indicates that while social influence is important, it is not the norm in the EC per se that determines intentions (Terry and Hogg 1996, Terry et al. 1999, Fielding et al. 2008). This result may indicate that other norms than those of the EC are connected to the intention to perform AES. In the farming community productivist norms are still widely held (Battershill and Gilg 1997, Mills et al. 2011, Burton and Schwarz 2013, McGuire et al. 2013). ECs have been hypothesized to create a group of likely-minded farmers, which could withstand these norms (Burton and Paragahawewa 2011, Burton and Schwarz 2013). However, our results do not provide support for this line of reasoning.

Perceived group pressure did not affect intention differently between farmers that identify strongly with the EC and those that do not. This contradicts previous results that farmers that identify strongly with the collective experience more group pressure and show a higher intention to perform conservation measures (Fielding et al. 2008). After personal contact with the coordinators of various ECs in our sample, they all confirmed that the ECs do not explicitly aim at exerting group pressure on their members to perform more AES; they merely play a facilitating role. This may explain why in our study we found no relation between intention and group norm, even for members that identify strongly with the EC. The significant relation between self-identity and the intention to perform meadow bird management highlights that if farmers consider meadow bird management more as part of their identity they intend more to protect meadow birds. Previous research found no effect of self-identity on farmers intention to perform subsidized AES, whereas it did for unsubsidized measures (Lokhorst et al. 2011) . Lokhorst explained this difference based on identity theory by stating that if people perform a certain behaviour voluntarily without any outside pressure or extrinsic reward, they see the behaviour as part of the self (Lokhorst et al. 2011, Lokhorst et al. 2013). For simple subsidized management the monetary reward provides the reason for performing this management, however, for unsubsidized practices where no subsidy covers the behavioural costs, the behaviour will be incorporated in the self-identity. In our study the difference in significance level of self-identity between ditch bank management and meadow bird management could be due to less behavioural and economic efforts (no fertilization or ditch slurry 1 metre outside of the ditch bank) of ditch bank management, while meadow bird management involves greater efforts (e.g. not mowing for some time, looking for nests in the fields). This may have led to the incorporation of meadow bird management into self-identity, whereas it did not for ditch bank management.

In our study, farmers' perceived profitability had no significant influence on their intention. This may seem striking as financial reasons have been listed by farmers as one of the most important reasons to enter in AES (Wilson and Hart 2001, Sutherland 2010). However, according to Sutherland (Sutherland 2011), farmers may over-emphasize financial reason in their reasoning to apply environmental measures, in order to present themselves as rational business people. Our findings underscore this argument.

We found great differences in the variables explaining the intention to perform either meadow bird management or ditch bank management, but also in the explained variation of intention. The difference we found in explained variation between ditch bank management (29%) and meadow bird management (69%) could be due to less behavioural and economic efforts (no fertilization or ditch slurry 1 metre outside of the ditch bank) of ditch bank management, while meadow bird management involves greater costs (e.g. not mowing for some time). It is assumed that the TPB is better in explaining behaviour that requires relatively high behavioural costs (Steg and Vlek 2009). Also, current ditch bank management is no longer applied collectively. Even though we inquired into ditch bank management which was performed up to 2009, opinions on current individual ditch bank management may have affected these variables.

Although our study's sample size was satisfactory, there are some limitations to the study design. Some predictors were measured with single items only, while others showed unsatisfying internal consistency resulting in single item measures as well.

Second, our data represent cross-sectional data, so we can describe our results in terms of associations and relations, not in causal effects. Third, we were unable to include a direct behavioural measure in our study, which means we only analysed self-reported intentions. Future research should pay special attention to collecting behavioural data, in the form of diaries or on-farm observations for instance, so that these can be integrated in the model. Additional to including behavioural measures, it would be particularly useful to for follow-up research to include ecological measures. This way, it can be assessed how the behaviour succeeds in achieving its goals; an increase in meadow birds and ditch bank plants.

Finally, both for meadow bird management and ditch bank management, a payment by result approach on collective management has been applied in the Netherlands between 1998 and 2009. In this system the EC applied for subsidy from the government and based on the collective funds from the subsidy, farmers were reimbursed based on the number of nests of meadow birds or the number of designated plant species per kilometre of ditch bank (Musters et al. 2001, van Dijk et al. 2013a). The EC served in this system as a regional agent to coordinate individual behaviour (Ferraro and Kiss 2002). From 2009 on the regional approach and payment by results for ditch bank management was abandoned. For meadow birds collective management is still performed in all ECs, but the payment by result was abolished by some. Two of the ECs in our study still apply a payment by result approach. Payment by result schemes have been hypothesized to enhance farmers' motivation for AES, by making the outcomes of management more visible to farmers. This would contribute to the generation skills and knowledge among farmers, but also to the exchange of these skills and knowledge between farmers, especially within ECs (Burton and Paragahawewa 2011, Burton and Schwarz 2013). This can have affected our results by for example increasing farmers perceived facilitation of the EC as compared to collective AES without payment by results. Further research would be required to study the influence of the payment by results approach on farmers motivation and how this changes in collective AES.

5. Conclusion

In conclusion, we found no relation between perceived norms within the EC and farmers' intention to perform AES. However, the extent to which farmers felt facilitated by their EC was positively associated with the intention to conserve. This shows that regional cooperatives have an added value in enhancing farmers' intention to perform AES and can be a valuable tool elsewhere to enhance farmers' willingness to perform AES.

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5. Factors underlying farmers' intentions to perform unsubsidized agri-environmental measures.

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Abstract

Over the last decades there is a growing body of literature on how to enhance farmers' participation in voluntary subsidized agri-environmental programmes. However, additional unsubsidized agri-environmental measures that farmers perform are often ignored. The willingness to perform these measures may give a better insight into farmers' motivation for agri-environmental measures than subsidized measures because it likely depends only on farmers' intrinsic motivation and not on extrinsic factors such as a financial compensation. In this study we used an extended version of the Theory of Planned Behaviour (TPB) to investigate which factors are associated with farmers' intention to perform unsubsidized agri-environmental measures. Our results demonstrate that attitude, perceived social norms and perceived personal ability control play a significant role in farmers' intention to perform these measures. However, self-identity is the most dominant predictor of farmers' intention to perform these measures. Furthermore we found that Environmental Cooperatives (ECs) positively influence farmers' willingness to perform additional unsubsidized measures by means of facilitation and group pressure. We conclude that in order to increase farmers' willingness to perform agri-environmental measures, self-identity should be addressed more by means of e.g. benchmarking instruments in combination with commitment making or labelling of environmental friendly identities. Also, ECs should be more involved in the performance of additional unsubsidized measures since they play a significant role in farmers' willingness to perform them, but so far ECs have been focussing mainly on subsidized agri-environmental measures.

1. Introduction

Over the last decades an unprecedented decline in biodiversity on farmland has taken place in Europe due to the intensification of agriculture (Stoate et al. 2001, Kleijn et al. 2009, Stoate et al. 2009). To halt this decline, two approaches have been applied: the founding of reserves and the implementation of subsidized agri-environmental measures in the form of agri-environmental schemes (AES) by farmers to protect biodiversity in agricultural areas. The top-down measures of founding reserves are a very costly solution, and have not always been successful in restoring species populations (Ferraro and Pattanayak 2006, de Snoo et al. 2012b, Jongeneel et al. 2012). The alternative, bottom-up voluntary AES are implemented in Europe and advocated abroad (Kleijn et al. 2006, Attwood et al. 2009, Shiki 2010). These schemes are alleged to provide a cheaper alternative to nature reserves (Jongeneel et al. 2012), but their effectiveness in conserving farmland biodiversity is still debated. (Kleijn et al. 2011, Kuiper et al. 2013, van Dijk et al. 2013a, van Dijk et al. 2014)

There has been a lot of empirical research on how to improve the effectiveness of currently applied subsidized agri-environmental measures, mainly from an ecological perspective. However, there is a growing awareness that nature conservation and specifically the conservation of farmland biodiversity may be a social challenge as well (Ferraro and Kiss 2002, Balmford et al. 2005, de Snoo et al. 2012a, Dramstad and Fjellstad 2013). Over the last years there has been a growing body of studies on social aspects of AES (de Snoo and Bertels 2001, de Snoo 2006, Lokhorst et al. 2010, Lokhorst et al. 2011, Burton and Schwarz 2013, Uthes and Matzdorf 2013). One main point of criticism that came out of these studies is that most AES currently fail to create a long lasting change in farmers attitudes and practice towards a more environmental friendly farming practice (Burton and Paragahawewa 2011, de Snoo et al. 2012a). This can be attributed to a lack of embedding agri-environmental measures into farmers' perception of what a 'good farmer' ought to do (Burton 2004, Burton and Paragahawewa 2011). The main reason for this is that the current incentive for farmers to participate in AES is assumed to be the financial reward. As such, farmers are regarded as acting solely with an economic rationale (Burton and Paragahawewa 2011). Although financial reasons have been listed by farmers as an important reason to participate in AES (Wilson and Hart 2001, Sutherland 2010), their importance may be overemphasized by farmers to represent themselves as rational business people (Sutherland 2011). Previous research has demonstrated that farmers make decisions on their farming practice also in a social and cultural context, in which factors like social pressure, self-identity and prestige play an important role (Burton and Wilson 2006, Fielding et al. 2008, Burton and Paragahawewa 2011, Lokhorst et al. 2011), which may even overrule financial motives (Chapter 4).

The financial focus of current AES does not only disregard the before-mentioned social determinants, it may even deteriorate farmers motivation to perform certain voluntary measures to protect the environment without financial reimbursement, by shifting their motivation from intrinsic ('a good farmer ought to protect nature') to extrinsic motivation ('I'm in it for the money'). This so called undermining effect in which intrinsic motivation is replaced by external rewards such as money has been documented since the 1970's; for an overview see (Deci et al. 1999). In the context of agricultural nature conservation, this effect has been documented in Finland, where the motivation for the traditional voluntary hanging of nest-boxes by farmers without compensation was replaced by the expectancy of a financial reimbursement to do so (Herzon and Mikk 2007).

Based on the importance of the social and cultural context of agri-environmental measures, one of the proposed measures to improve current AES is organizing farmers in regional cooperatives by means of collective AES (Franks and Mc Gloin 2007, Mills et al. 2011, Burton and Schwarz 2013, Franks and Emery 2013). These cooperatives are hypothesized to create a peer-group of farmers who all perform agri-environmental management. These cooperatives can create social norms to perform agri-environmental measures, possibly resisting productivists norms, which are still very salient among the farming community (Battershill and Gilg 1997, McGuire et al. 2013). In the Netherlands, where collective AES have been implemented since the beginning of 2000, the so called Environmental Cooperatives (EC) have been demonstrated to facilitate farmers in the performance of subsidized AES (Chapter 4). The benefits of the concept of Dutch collective AES in ECs have resulted in the inclusion of AES at a regional level in the current reforms of the European Common Agricultural Policy after successful pilots in the Netherlands (European Commission 2011, IEEP 2012). Previous research has yielded mixed results on the influence that group norms of these cooperatives have on farmers' intentions to perform agri-environmental measures. In some studies group norms had a positive effect if farmers identified with the group (Beedell and Rehman 2000, Fielding et al. 2008) but in other studies no effect of group norms was found (Chapter 4).

There has been a growing demand to make voluntary agri-environmental measures such as AES more socially sustainable among farmers by creating a long-lasting change in farmers' motivations and measures to protect the environment. So far, the body of research from a social science perspective on farmers performing voluntary subsidized agri-environmental measures with financial rewards has grown steadily over the last years (Franks and McGloin 2007, Sutherland 2010, Burton and Paragahawewa 2011, Emery and Franks 2012, Mills 2012, Burton and Schwarz 2013, Franks and Emery 2013). However, the social aspects of farmers performing additional environment-beneficiary measures without a financial reward is studied only to a limited extend.

In a study where a direct comparison of motivational factors between agri-environmental measures with and without financial compensation found great differences (Lokhorst et al. 2011).

First, internal motivational factors explained a far greater amount of intention to perform unsubsidized agri-environmental measures than subsidized measures. Second, variables like self-identity and social pressure played a significant role in unsubsidized management, whereas they did not in subsidized management. Self-identity reflects to what extent a person sees the performance of a certain behaviour as part of fulfilling a certain societal role which is part of the self, for example a farmer performs unsubsidized agri-environmental management when (s)he sees him or herself as a farmer who is concerned with the environment" (Conner and Armitage 1998). If the performance of these measures is mainly driven by self-identity rather than financial compensation, it becomes less likely that this behaviour is easily affected by for example changes in financial policies e.g. subsidized AES. Finally, previous research has demonstrated that membership of an EC is positively related with farmers' intention to perform subsidized agri-environmental measures (Chapter 4). So far the influence of these collectives on farmers' intention to perform unsubsidized measures has not been studied. This indicates that unsubsidized agri-environmental measures may provide a better starting point into what motivational factors may alter the social sustainability of agri-environmental measures than does research on subsidized measures.

In this study we will investigate what motivational factors underlie farmers' intention to perform unsubsidized agri-environmental measures and how this results in the actual performance of these measures. We will specifically study how the membership of an EC affects farmers' intention to perform unsubsidized measures. As far as we know, this is the first study to investigate how the performance of collective AES in EC affects the intention and performance of additional unsubsidized agri-environmental measures by farmers.

1.1. Our model

To investigate the effect of membership of an EC, we use the Theory of Planned Behaviour (TPB)(Fig. 1)(Ajzen 1991). This theory states that the performance of a certain behaviour is directly dependent upon the intention one has to perform this behaviour. Intention, in turn, depends on three factors; the attitude towards the behaviour, the perceived social pressure from significant other people to perform the behaviour (subjective norm) and perceived behavioural control (PBC), which encompasses the perceived ability to

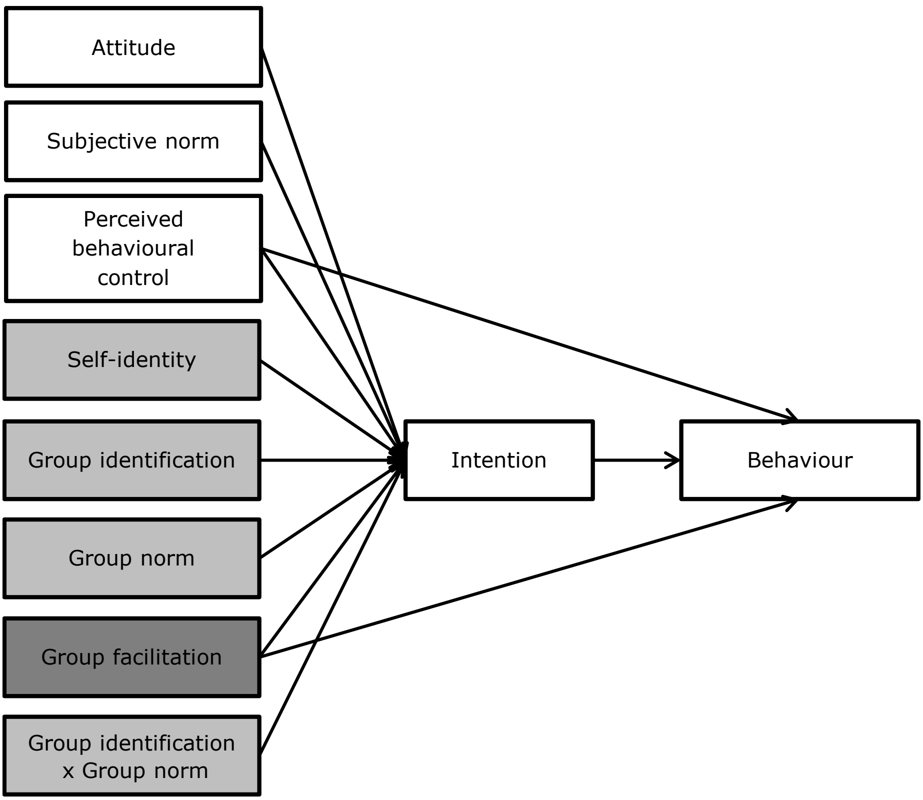

Figure 1 Graphical representation of our expanded theory of planned behaviour .
The white square represent the traditional TPB items (Ajzen 1991), light grey squares represent the additional items based on social identity theory (Terry et al. 1999) and the dark grey items represent our additions based on group facilitation.

perform the behaviour. The TPB has been advocated and demonstrated to provide a good theoretical framework to explain land-owners decisions to perform agri-environmental measures (Burton 2004, Fielding et al. 2008, Sutherland 2010, Wauters et al. 2010, Lokhorst et al. 2011, Mastrangelo et al. 2013).

Next to the traditional TPB variables we extended our model with the predictor self-identity. Self-identity has been demonstrated to be a significant contributor to intention, in addition to the standard TPB items. Specifically, self-identity has been shown to be significantly associated with intention to perform agri-environmental measures (Lokhorst et al. 2011, Mastrangelo et al. 2013), and its influence seems to increase with more costly agri-environmental measures (Burton 2004, Lokhorst et al. 2011, Chapter 4).

Second, to include the effect of perceived group pressure that other members of the EC may perform, we will include two other variables derived from social identity theory; group norm, which measures perceived social pressure from the EC, and identification with the EC (group identification) (Terry et al. 1999, Fielding et al. 2008) (Chapter 4). Previous research has demonstrated that group pressure may alter individuals intention to perform a certain behaviour, particularly for individuals that identify strongly with the EC (Terry et al. 1999, Fielding et al. 2008).

Finally, we will include a variable to measure the facilitating role of the EC in performing unsubsidized agri-environmental measures. The facilitation of subsidized AES by ECs has been widely discussed in the literature (Glasbergen 2000, Wiskerke et al. 2003, Franks and McGloin 2007), yet some ECs also help with unsubsidized projects. This facilitation could be included in the measure for perceived behavioural control, however, the reliability of PBC in teasing apart the perceived personal ability and the facilitation by external factors has been questioned (Conner and Armitage 1998, Manstead and van Eekelen 1998, Ajzen 2002). Therefore we will add the variable group facilitation, which measures the facilitation by the EC (Chapter 4). This study is the first to provide empirical insights in whether Environmental Cooperatives alter farmers’ intention to perform unsubsidized agri-environmental measures, resulting in the performance of more measures.

We hypothesize, in accordance with the TPB model, that attitudes, subjective norms and perceived behavioural control are positively associated with the intention to perform unsubsidized agri-environmental measures (H1). Based on the results of Lokhorst et al. (2011) and Chapter 3 we expect self-identity to explain a significant amount of variation in the intention to perform unsubsidized measures (H2). Some ECs are involved with unsubsidized practices, therefore we expect group norm to have a main effect on intention for unsubsidized measures (H3). Particularly, for farmers that identify strongly with the EC we expect a positive effect of group norm on intention to perform unsubsidized management (H4), in line with Fielding (2008). Some ECs help with unsubsidized projects, therefore we expect that farmers perceive an increased ability to perform unsubsidized measures because of group facilitation by the EC (H5). Finally, in line with the TPB model, we expect intention and perceived behavioural control to explain behaviour, but we also expect group facilitation to contribute significantly to the behaviour (H6).

2. Material and methods

2.1. Procedure and Participants

To test for a comprehensible phrasing of the questions, we tested a first version of the questionnaire with 5 different farmers. We sent 720 surveys by mail to dairy farmers who are member of 5 ECs in the Western part of the Netherlands. To increase the response rate we added a meadow bird identification poster and after 2 weeks a reminder email was sent. Participants who had not returned the questionnaire after 2 months were called personally to ask if we could resend a survey. Finally, a total of 384 surveys were returned, representing a 53.3% response rate. Out of these a total of 314 farmers filled out the questionnaire appropriately. The median age of these farmers was 45 years (M=48.8, range:25-75) with a mean farm size of 41 hectares (M=36.5, range: 1-200 ha), which is a bit higher than the average size of dairy farms in the Netherlands of 30 hectares (LEI 2013).

2.2. Questionnaire

To investigate the number of unsubsidized agri-environmental measures that farmers performed we first made unsubsidized measures comprehensive and salient by inquiring into how many agri-environmental measures farmers performed on their farm of a list of 23 possible measures. Examples of measures are for instance the presence and maintenance of hedges, truncated trees, thickets, nesting boxes for songbirds, nesting poles for storks, haystacks in the meadows and stacks of branches or bricks. We also asked whether these measures were subsidized or unsubsidized. As an indicator for behaviour we used the number of measures performed of the list of 23 agri-environmental measures. We then standardized the number of measures to fit them in a 5-point Likert scale.

Subsequently we enquired into the TPB items based on the ticked landscape measures that farmers perform. To obtain a response rate as high as possible we used a modified scale with fewer items. For all items we used a 5-point Likert scale.

To measure intention we used two items (Perugini and Bagozzi 2001): “In the future, under the same conditions I would perform unsubsidized agri-environmental management again” and “Given the opportunity I would perform unsubsidized agri-environmental management again: certainly not / certainly”. Cronbach’s α for this scale was 0.82. Attitude was measured using the items “I think that unsubsidized agri-environmental management is: negative / positive”, “I think unsubsidized agri-environmental management is: useless / useful”, and “I think that unsubsidized agri-environmental management is: unimportant / important” (Ajzen and Fishbein 1980). This scale did not yielded a sufficiently high reliability (α=.67). Therefore a single item operationalization was used; “I think that unsubsidized agri-environmental management is: useless / useful” in our analysis. For subjective norm we used the item “Most people who are important to me think it is important that I carry out unsubsidized agri-environmental management: completely disagree / completely agree” (Ajzen and Fishbein 1980).

Perceived behavioural control (PBC) was measured using the following item “I think it is difficult to carry out unsubsidized agri-environmental management” (Ajzen and Fishbein 1980).

We used two items to measure self-identity: “Unsubsidized agri-environmental management is part of who I am” and “Unsubsidized agri-environmental management is something that is typical for me” (Terry et al. 1999, Lokhorst et al. 2011). For this scales we obtained Crohnbach’s α of 0.73.

To measure the group norms of the EC we used the item “Most members of my EC think it is important that I carry out unsubsidized agri-environmental management” (Chapter 4).

Group facilitation was measured using the item “ Thanks to the EC, I am capable of carrying out unsubsidized agri-environmental management: completely disagree / completely agree” (Chapter 4).

To measure identification with the EC we used four items (Mael and Ashforth 1992); “I am affected if members of my EC are being criticized”, “If members of my EC are thriving I am happy”, “I am proud of the members of my EC” and “When I’m talking about the members of my EC, I’m talking about “we” and not “them”. For these scales we obtained Crohnbach’s α of 0.80.

2.3. Statistics

We performed a structural equations model (SEM) using AMOS 21 (IBM, Chicago, IL, USA) to our model to test for the significance of the variables on intention and behaviour (Fig. 1). To test for a good model fit we use two goodness of fit indices: the Comparative-Fit-Index (CFI), the Standardized-Root-Mean-Square-Residual (SRMR) and the Root-Mean-Square-Error-of-Approximation (RMSEA). The cut-off values applied for a satisfactory model fit were a RMSEA below 0.08, a SRMR below 0.08 and a CFI of at least 0.90 in line with the traditional recommendations (Hu and Bentler 1999).

3. Results

From the list of 23 measures that we inquired into, farmers performed on average 6.6 measures. The three most popular measures were making stables accessible for birds such as swallows and barn owls (93%), maintaining rows of trees (61%) or individual trees (59%) and the presence of stacks of branches or tree trunks (55%). The least popular measures were the presence of a bat house (7%), pond (6%) and nesting pole for storks (4%).

Table 1 Means, standard deviations and correlations for measures.

		M	SD	1	2	3	4	5	6	7	8
1	Behaviour	2,14	0,65	1							
2	Intention	3,48	1,05	,19**	1						
3	Attitude	3,38	1,27	,16**	,58**	1					
4	Subjective norm	3,00	1,13	,17**	,45**	,33**	1				
5	PBC	3,36	1,19	,12*	,35**	,35**	,07	1			
6	Self-identity	3,34	1,00	,18**	,72**	,55**	,43**	,27**	1		
7	Group norm	2,90	1,07	,19**	,36**	,19**	,52**	,00	,39**	1	
8	Group facilitation	2,58	1,19	-,06	,25**	,10	,29**	,01	,22**	,36**	1
9	Group identification	3,67	0,81	,01	,11	,01	,20**	-,05	,20**	,22**	,22**

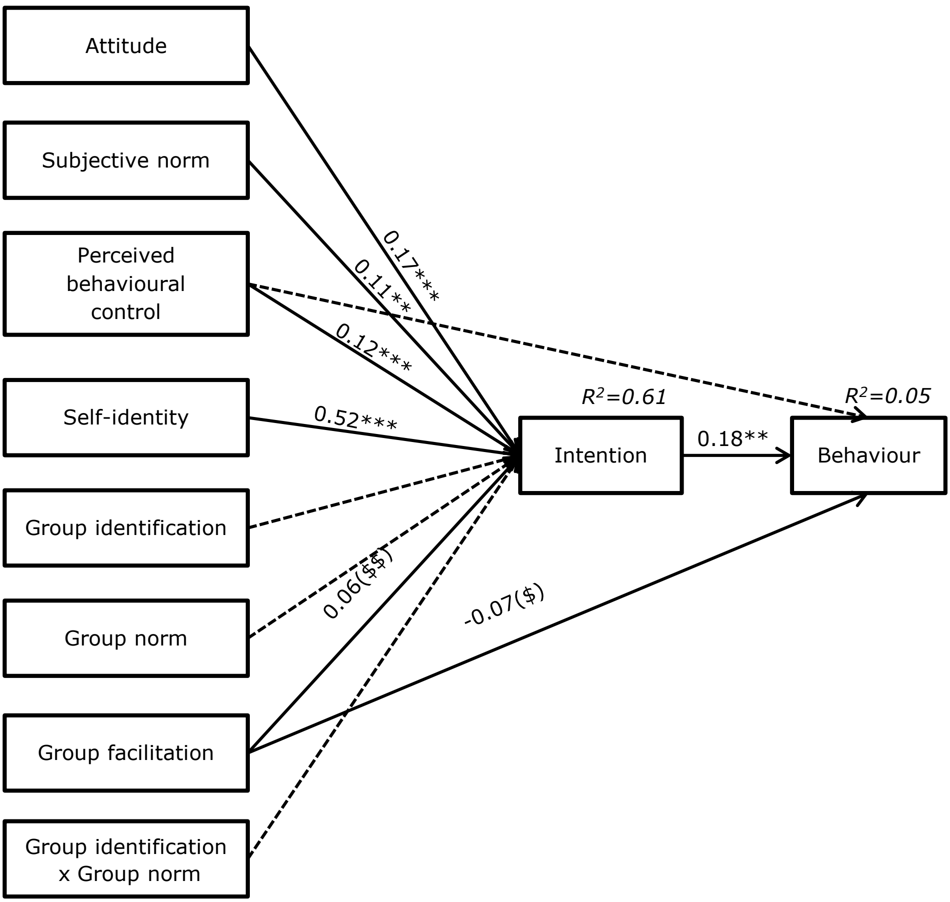

Figure 2 Results of the structural equations model for the extended theory of planned behaviour. Solid lines represent significant relations, dashed lines are non-significant relations. Significance levels: ($$)P= 0.082, ($)P= 0.054 **P<0.01, ***P<0.001

The results from the TPB items demonstrate that intention and behaviour are correlated, in line with the assumptions of TPB (Table 1). Furthermore, in line with the assumptions of TPB, are all variables correlated to intention, except for group-identification. The strongest correlations are between self-identity and intention, followed by attitude and intention. Group norm correlated strongest with subjective norm. The average values demonstrate that farmers on average do not perceive strong group norms or facilitation from the EC.

The model fits of the SEM (RMSEA: 0.077, CFI: .982, SRMR:0.029) indicate an acceptable fit of our model. The three standard TPB items attitude, subjective norm and PBC all explain a significant amount of variation in intention to perform unsubsidized measures, confirming Hypothesis 1 (Fig. 2). However, self-identity has the largest b of all variables (b=0.517, t: 10.392, P<0.001), which confirms Hypothesis 2. Of the variables measuring the influence of the EC only group PBC explained a marginally significant amount of variation in intention, (b=0.059, t: 1.715, P=0.087) but also in the behaviour (b=-0.060, t:-1.920, P=0.054). Remarkably, in the bivariate analysis, group facilitation and behaviour were not correlated (r=-0.06, P=0.290, see Table 1). The correlation between group facilitation and intention (r=25, P<0.01) may have resulted in a confounding effect. To test this we repeated our model without intention and to test the independent effect of group facilitation. In this model, group facilitation had no significant association with behaviour (B=-0.38, t=-1.092, P=0.276). Next we repeated our model without group facilitation. In this model, intention was still significantly positively associated with behaviour (B=0.121, t=2.849, P<0.01). This demonstrates that even without group facilitation, intention is significantly associated with behaviour. Group facilitation, on the other hand had no significant association with behaviour if intention is not included in the model. Only if intention was included, the confounding effect between group facilitation and intention resulted in a marginally significant association of group facilitation with behaviour.

Group norm, which in the bivariate analysis correlated stronger with intention than group PBC (resp. r:0.36 & r:0.25), had no significant effect on intention (B=0.048, t=1.111, P=0.267). Because group norm is both positively correlated with intention and subjective norm (r:0.52) but has no significant effect in our final model, mediation via subjective norm is likely. To test this we first repeated our model, but without subjective norm. In this model group norm was significant (B=0.093, t=2.313, P<0.05). To test for mediation of subjective norm on group norm we performed a bootstrapping analysis (Preacher and Hayes 2004). The bias corrected bootstrap estimate of the indirect effect of group norm had a 95% confidence interval of 0.113 to 0.263. This confidence interval suggests a significant mediating effect of group norm on intention via subjective norm.

Our results confirm Hypotheses 1 and 2 that the traditional TPB constructs attitude, subjective norm and perceived behavioural, and the addition self-identity play a significant role in the intention to perform unsubsidized agri-environmental measures.

We initially found no direct effect of group norm on intention, however there is an indirect effect of group norm via subjective norm on intention, which confirms hypothesis 3. No significant difference was obtained in group norm between farmers that identify differently with the EC, rejecting hypothesis 4. We found a marginally significant effect of group PBC on intention, therefore we can confirm hypothesis 5. In line with TPB, intention explained a significant amount of variation in actual behaviour, but neither PBC, nor group PBC had a significant effect on behaviour, rejecting hypothesis 6.

4. Discussion

Studies on farmers' motivation to perform agri-environmental measures usually focus on the performance of subsidized agri-environmental measures such as AES (Wilson 1997, Wilson and Hart 2001, Burton and Paragahawewa 2011). However, voluntary unsubsidized agri-environmental measures may provide a better starting point to investigate farmers' motivation, because they depend solely on the farmers' intrinsic motivation rather than extrinsic factors such as subsidies, which are vulnerable to changes in financial policies and political climate.

The study at hand aligns with previous research on farmers' motivation for nature conservation in several ways. First, by stressing the importance of self-identity as a predictor for the intention to perform agri-environmental measures and the actual performance of agri-environmental measures. Our findings corroborate previous research which demonstrated that self-identity played a significant role for farmers to perform unsubsidized measures whereas it did not for subsidized measures (Lokhorst et al. 2011). Yet in an earlier study it was found that within subsidized measures the effect of self-identity varied per type of AES; a significant positive effect for the protection of meadow birds but no effect for the management of ditch banks (Chapter 4). The difference in the effect of self-identity between different kinds of agri-environmental measures may lie in the higher behavioural costs of for example unsubsidized measures where no financial compensation is given in return for the efforts. Lokhorst (2011) explained this difference based on self-perception theory, which states that people construct their self-image based on their actions rather than the other way around (Bem 1972). If someone performs a certain behaviour voluntarily and there is no obvious extrinsic reason to do so, a person will see himself as “the kind of person who does this”, leading to the incorporation of the behaviour in the self. This seems to be the case with unsubsidized management, where no obvious extrinsic reason, such as subsidy, can be used as rationale for the behaviour.

Based on previous research on different kinds of subsidized agri-environmental measures (Chapter 4) and our result that self-identity is the most influential variable on intention for unsubsidized measures, the current study indicates that the higher the behavioural costs a certain agri-environmental management involves, the higher the influence of self-identity.

A second important finding that extends previous research was that we found a marginally positive relation between facilitation by the EC and farmers intention to perform AES. However, farmers scored facilitation by the EC lowest compared to other predictors, suggesting room for improvement (Table 1). Nowadays ECs do not focus primarily on unsubsidized measures, however they sometimes facilitate some voluntary projects (Franks and Mc Gloin 2007). Our results indicate that if ECs would facilitate more unsubsidized measures, that this would result in an increase in uptake by members.

Third, contrary to our expectations, we did not find an association between group norm and intention independent of subjective norm. Neither did we find a significantly stronger association between intention and group norm for farmers that identify strongly with the EC. However, for group pressure we found a significant association with intention mediated by subjective norm. The mediation of subjective norm between intention and group norm contradicts the work of Terry and colleagues (Terry and Hogg 1996, Terry et al. 1999, Fielding et al. 2008) which showed an independent effect of group norms on intention in addition to the effect of subjective norm. How can this difference be explained? We believe this lies in the particular history of environmental cooperatives. The emergence of ECs in Netherlands dates back to the early nineties and the performance of collective AES to 2000 (Franks and McGloin 2007). The long lasting association between the farmers in our study and their fellow EC members may have resulted in an incorporation of this group into the “significant others” that determine the subjective norm, resulting in a mediation effect.

The influence of group norms and the facilitation by the EC empirically confirms the hypothesized advantages of collective AES; namely support and group pressure (Burton and Paragahawewa 2011, Mills et al. 2011, Franks and Emery 2013).

4.1. Limitations

Although we had a satisfying sample size and response rate, and our farmers seem to be representative for the average dairy farmer in the Netherlands, there are some limitations to our study design. First, our farmers may consist of front-runners in the area of agri-environmental measures, since all of them currently perform subsidized AES. This may have affected the representativity of the individuals included in our sample for the population. Second, some predictors were measured with a single item operator, while others lacked internal consistency resulting in single item measures as well. Finally, our study employs a cross-sectional design, meaning we measured correlative relations, not causal effects. Finally we measured behaviour and intention contemporaneously. This is problematic because it might not reflect the behaviour resulting from intention, but rather behavior performed in the past (Armitage and Conner 1999).

4.2. Implications

Current Europe-wide implemented agri-environmental measures vary widely in their effectiveness to preserve biodiversity. To improve this, more elaborate measures are necessary. However, this will require farmers' willingness to perform these.

Therefore, it becomes more and more important to understand why farmers might or might not be willing to adopt environmental measures. The current study adds to solving that puzzle by by demonstrating that self-identity is the most important factor to address to improve farmers' willingness to perform unsubsidized voluntary measures. How should our results be put into practice?

One approach to have farmers perform more agri-environmental measures is to apply benchmarking (de Snoo 2006, de Snoo et al. 2010). Benchmarking gives a frame of reference of the number of agri-environmental measures that a certain farmer performs compared to e.g. fellow EC members. This way, benchmarking provides farmers with a descriptive norm: a perception what other people do. People tend to conform to such norms (Cialdini et al. 1991). Our results demonstrate that norms from the EC (group norm) and from important other (subjective norms) may alter farmers' intention to perform AES, however they are not the most effective contributor. Previous research confirmed this, by demonstrating that benchmarking alone did not alter farmers' intention to perform more agri-environmental measures when compared to fellow EC members. However, benchmarking and commitment making did (Lokhorst et al. 2010). Commitment making is a psychological instrument which has been hypothesized to change a person's self-identity (Lokhorst et al. 2013). This implies that commitment making may be an effective instrument to increase the amount of agri-environmental measures that farmers perform (Lokhorst et al. 2010).

A second method to address self-identity is by emphasizing a certain identity by means of labelling. Labelling is emphasizing a certain identity by using a positive trait label (e.g. responsible nature-friendly farmer) while referring to a certain behaviour that a person performs. This technique has even been demonstrated to be able to counterfeit the undermining effect that extrinsic factors, such as a reward, can have on intention (Cialdini et al. 1998) by shifting the motivation from driven by the reward to self-identity. In the case of AES this could be done by labelling farmers that perform subsidized AES as responsible farmers; "You perform voluntary AES, this shows you are a farmer that cares about the protection of nature". This way the behaviour, protecting nature, is linked to a certain identity instead of the extrinsic financial reward. Using subsequently instruments that address this identity may eventually even lead to the performance of additional voluntary measures, since self-identity plays a major role in farmers' intention to perform these. This could be done for example by campaigns that address this self-identity "You might be interested in hanging a nesting box for swallows because you seem like a farmers that cares about nature."

Besides self-identity also the group facilitation by the EC can be addressed to increase the performance of additional agri-environmental measures. Group facilitation scored lowest compared to the other predictors whilst being associated with intention. Having ECs actively involved in more unsubsidized projects may lead to an increased uptake of these measures. ECs have been demonstrated to help farmers overcome barriers in performing AES by for example the sharing of knowledge and skills within the EC leading to an increase in willingness to perform agri-environmental measures (Franks and McGloin 2007, Chapter 4). This has even been hypothesized to affect self-identity because the participation of farmers in collective measures may reinforce their self-identity as conservationists, again underscoring the importance of the EC (Deaux and Martin 2003).

5. Conclusion

We conclude that self-identity is the most important determinant of farmers' intention to perform unsubsidized agri-environmental measures and should be addressed more to increase farmers willingness to perform additional measures to conserve biodiversity. Furthermore, the performance of agri-environmental measures in cooperatives like the ECs influences positively farmers intention to perform agri-environmental measures, but these ECs should be involved more into these measures.

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6. Can we improve agri-environment schemes by integrating ecology and psychology? A synthesis!

1. Introduction

Agri-environment schemes (AES) are one of the key policy instruments implemented in Europe to restore the declining biodiversity on farmland (Burton and Wilson 2006). One of the most commonly applied AES in Europe is the management of ditch banks neighbouring agricultural fields to conserve the biodiversity in these banks. However, field studies suggest that there is very little to no increase in the number of species in ditch banks with AES (Kleijn et al. 2001, Blomqvist et al. 2009, Leng et al. 2011). This has led to more and more authors starting to question the efficacy of ditch bank management in achieving an increase in the number of plant species in ditch banks over the last 10 years. In this thesis I have addressed the factors that may underlie the current ineffectiveness of ditch bank management using a large scale, long term dataset collected by farmers. I analysed both ecological and socio-economic factors to address the improvement of AES as both an ecological and social challenge (de Snoo et al. 2012a).

1.1. Trends in plant composition in ditch banks

In Chapter 1 I have demonstrated that the target species richness in ditch banks has slightly increased over the last 10 years. However, I found no difference in the number of target species between ditch banks with and without AES. These results are in line with the previous small scale study on the effectiveness on ditch bank management (Blomqvist et al. 2009). To investigate the ecological processes underlying these trends, I characterized the vegetation based on traits related to seed- and site-limitations (Blomqvist et al. 2006).

1.2. Site-limitations

To investigate more in depth the effect of site-limitation on species richness I calculated in Chapter 1 the trend in the average nitrogen demand of the observed target species using their Ellenberg N values. This resulted in a slight increase in the Ellenberg N value between 2000 and 2009 indicating that species that have a high tolerance of nitrogen are slightly increasing. These results give weight to field studies that have questioned the effectiveness of currently applied measures aiming at decreasing the nutrient supply (Kleijn et al. 2001, Kleijn et al. 2004, Blomqvist et al. 2009). The increase of high nitrogen tolerant species reflects an advantage for tall, fast growing species compared to the slow growing, short plant species typical of hay meadows that are aimed for (Blomqvist et al. 2009, van Dijk et al. 2013b). In order to make ditch bank management more effective in enhancing the species richness in ditch banks, we recommend more elaborate measures that really lead to a decrease in productivity and nutrient supply in ditch banks (Blomqvist et al. 2006, Blomqvist et al. 2009).

1.3. Seed-limitations

Another process that may hamper the outcomes of ditch bank management is the limited availability of seeds. In Chapter 1 I characterised the observed species based on their seed dispersal vector. The slight increase in target species richness I found could be contributed to species that are able to disperse over long distances by water, while species that had no capacity to disperse over long distances declined. These results indicate that vegetation changes in AES ditch banks are affected by limitations in the dispersal of seeds (Blomqvist et al. 2006).

In Chapter 2 I followed up on this result by investigating the long term effects of seed dispersal from species-rich sources, such as nature reserves in the vicinity of AES. My results showed that the species richness was higher in ditch banks close to nature reserves, compared to ditch banks at greater distances. I found that this trend flattened out in time due to an increase of target species with increasing distance from nature reserves. However, this process appears to be limited to a few species, mainly those that disperse by water, such as Iris pseudacorus (Leng et al. 2010b). Species with no adaptation to disperse over long distances, such as Silene flos-cuculi, are declining even in the vicinity of nature reserves. The small effect of time and distance on species richness that I found shows that the process of dispersal from species rich sources is either very slow or limited to only a few species. I concluded that ditch banks can have the potential to serve as corridors by which species can disperse from species rich sources throughout the landscape but proper spatial planning taking nature reserves into account is required in order to take advantage of this potential (Soons et al. 2005, Leng et al. 2010b, Soomers et al. 2010, de Snoo et al. 2012a).

2. How to implement more elaborate measures?

In Chapter 1 and 2 I have concluded that more elaborate measures are required at locations with a favourable landscape configuration. This raises the question how policy makers can implement these recommendations. There are two possible ways: adapting the prescriptions of currently subsidized AES or motivating farmers to perform additional measures in their ditch banks without being paid for it. In both approaches the willingness of farmers to change the management is critical. Previous research has demonstrated that for these two approaches different motivational factors underlie farmers' willingness to adjust their management (Lokhorst et al. 2011). Therefore I have investigated in Chapter 3 the motivational factors underlying farmers' intention to perform subsidized AES and in Chapter 4 the factors underlying unsubsidized environmental measures.

2.1. Subsidized environmental measures

I analysed in Chapter 3 what variables influence farmers' intention to perform subsidized ditch bank management using an extended version of the Theory of Planned Behaviour (Ajzen 1991). Because I expected that the collective performance of ditch bank management in ECs may affect farmers intention I included 2 variables measuring how facilitation and group pressure from the EC affects farmers' intention. To compare between different kinds of collective AES I also analysed the effects of these factors for meadow bird management schemes. The results demonstrated that attitude and perceived behavioural control were significantly associated with farmers' intention to perform ditch bank management. This means that farmers' beliefs of the expected outcomes of ditch bank management and their expected ability to successfully perform the management determines whether a farmer intends to perform ditch bank management or not. I found no association between facilitation or group pressure by the EC and the farmer's willingness to perform ditch bank management.

As comparison, for meadow bird management schemes I also found that attitude and Perceived Behavioural Control (PBC) had the strongest association with intention.

However, the association between attitude and intention was weaker than for ditch bank management. Furthermore, I found a significant relation between facilitation by the EC and farmers' intention to perform meadow bird management. Also subjective norm and self-identity were significantly associated with farmers' intention to perform meadow bird management. This indicates that the extent to which farmers perceive that important other people expect them to protect meadow birds, and the extent to which they see themselves as people who protect meadow birds is related to their willingness to protect meadow birds, whereas this is not the case for ditch bank management. The results for ditch bank management agree with previous research results concerning field margin management on arable farms that demonstrated that farmers' intention for subsidized agri-environmental measures was only associated with attitude (Lokhorst et al. 2011), whereas for meadow bird management, more factors affect farmers' intention.

2.2. Unsubsidized environmental measures

In Chapter 4 I investigated which factors determine farmers' willingness to perform certain voluntary unsubsidized environmental measures. These measures were not part of ditch bank management but incorporated for example the maintenance of landscape elements such as trees, hedges or the hanging of nesting boxes for birds. I expect that these factors also provide insight into farmers' intention to perform additional unsubsidized measures in ditch banks.

In line with Chapter 3 we used the TPB but we could extend the model with a behavioural scale in the form of the number of unsubsidized environmental measures performed. Our results demonstrated that in addition to subsidized measures, farmers perform a wide arrange of voluntary measures. Contrary to ditch bank management and meadow bird management, I found self-identity to have the strongest association with intention, in line with previous results (Lokhorst et al. 2011). Attitude and PBC were significantly associated with intention, but the association was weaker than for ditch bank and meadow bird management as found in Chapter 3. Furthermore we found that the EC played a significant role by means or facilitation and group pressure, although unsubsidized measures are not the primary objective of ECs. These results indicate that in order to enhance the willingness of farmers to perform additional measures without financial compensation, other variables play a role than in an approach by which farmers receive subsidy. This difference is mainly reflected in an increased association between intention and self-identity.

3. What measures do farmers perform?

In this thesis I recommend that more elaborate measures are required to increase the plant species richness and give insight into two approaches how to motivate farmers to perform these measures. However, a few questions remain.

In Chapter 1 and 2 I assumed that farmers that participated in ditch bank management performed only the measures prescribed in the AES. However, the payment by results approach of the Dutch ditch bank management scheme has been hypothesized to encourage farmers to take additional measures besides the management prescriptions (Musters et al. 2001, Burton and Schwarz 2013). To investigate what subsidized and what additional unsubsidized measures farmers perform I used the questionnaire of Chapter 3 to inquire about the performance of 9 possible measures in ditch banks. These measures consisted of 2 mandatory prescribed measures for subsidized ditch bank management:

1) no fertilization and 2) never depositing ditch bank slurry in the outermost metre of ditch bank (DLG 2000). Furthermore I inquired about 7 additional voluntary measures such: avoiding fertilization of the outermost 2 metres of ditch bank, avoiding the use of pesticides on ditch banks, direct removal of hay from ditch banks, only mowing the ditch banks between half July and September, no mowing at all, no frequent cultivation, and fencing off of the ditch bank.

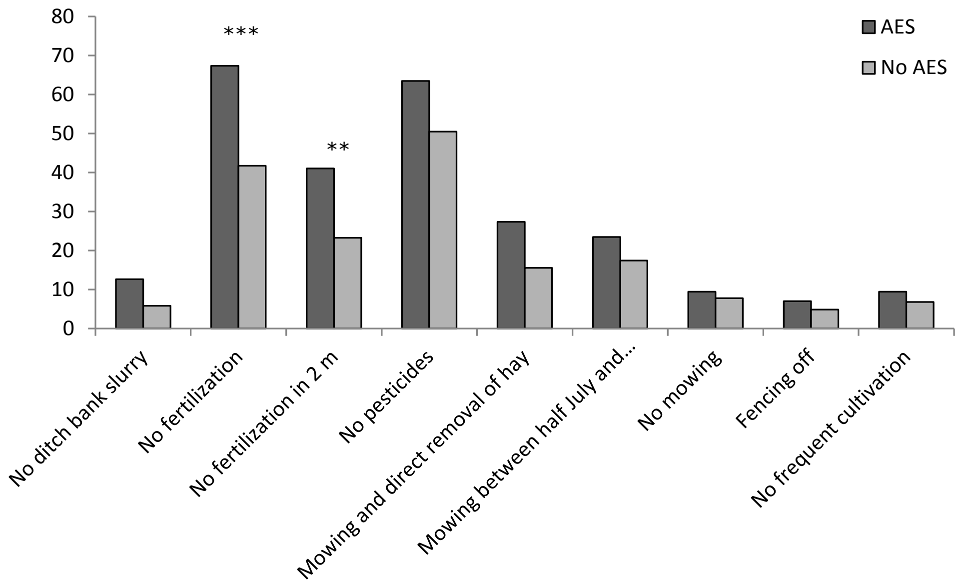

Figure 1 Measures taken by farmers with and without subsidized AES in their ditch banks in the period 2000-2009 out of a total of 297 farmers questioned. No ditch bank slurry and no fertilization are mandatory measures under AES, the other measures are voluntary. Significance values based on Chi-square test per measure with a Yates' continuity correction. ***:P<0.001, **:P<0.01.

Figure 1 demonstrates the percentage of farmers with and without subsidized ditch bank management performing the 9 individual measures inquired into. Remarkably, I found only for no fertilization a significant difference between AES and no AES (X^2=20.138, P<0.001). Of the 7 additional voluntary measures I only found a significant higher frequency of no fertilization in the outermost 2 meters of ditch bank (X^2=6.82, P<0.01) for farmers that received subsidy. The additional 2 meters of ditch banks without fertilization may be an artefact because ditch bank management from 2010 on was extended to 2 instead of 1 metre outside of the water surface of the ditch. Although I inquired about ditch bank management between 2000 and 2009, current management practice may have influenced farmers' responses about what they did in the past. The results show that farmers perform different measures than as prescribed by the management prescriptions. This raises the important question: What are the effects of the actual performed measures on target species richness in ditch banks?

4. How do the measures affect the target species richness?
To obtain the effect of the performance of the measures in Figure 1 on the number of target species, I have linked in this synthesis the data of Chapters 3 and 4 with the monitoring data of target species in 2009 from Chapter 1. I was able to connect the questionnaire data with the monitoring data of target species in 2009 for 157 farmers. There were differences between the target species monitored between ECs, therefore I used only the 16 target species that were monitored in all ECs. I used the average number of the 16 target species per farmer as our species scale.

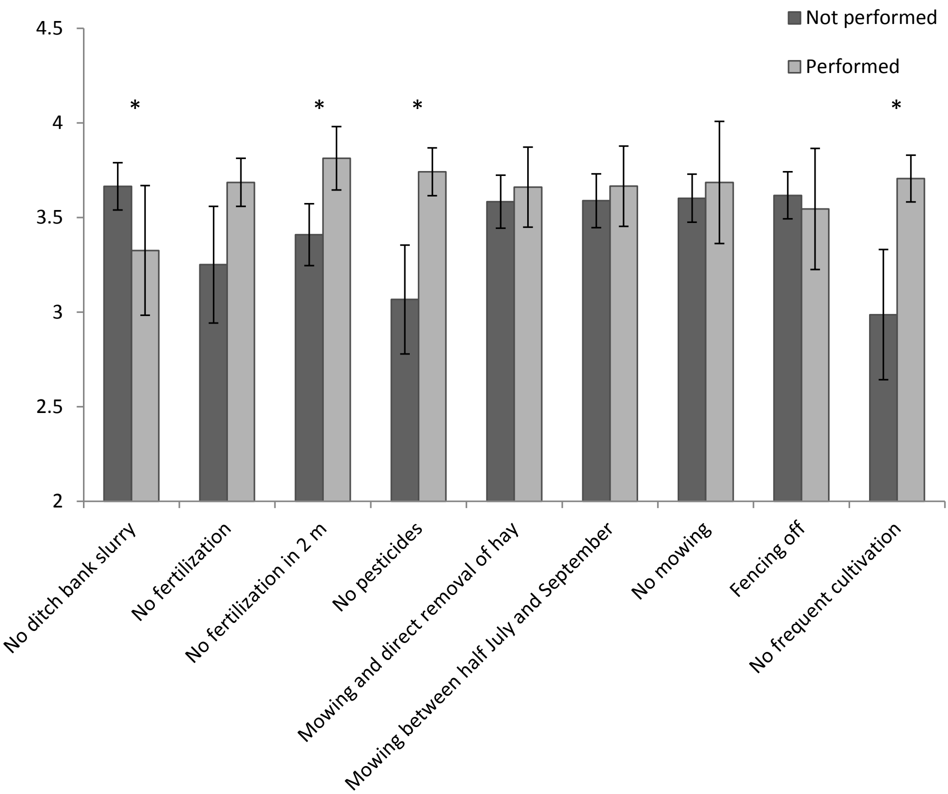

Figure 2 Number of target species in 2009 of 157 farmers with subsidized ditch bank management performing certain measures in their ditch banks between 2000 and 2009. No ditch bank slurry and no fertilization are mandatory measures under AES. Significance values based on an ANOVA with average target species number as dependent variable and the performance of each of the 9 individual measures as independent variables. *:P<0.05. Error bars indicate standard errors.

Remarkably, Figure 2 shows that of the mandatory 2 measures of subsidized ditch bank management, no fertilisation in the outermost meter had no effect on target species richness and no deposition of ditch slurry in the ditch bank had a significant negative effect on the number of target species (B:-0.72, $F_{1,121}$:4.44, P<0.05). The latter may be because the deposition of ditch bank slurry helps both with the dispersal of floating seeds from the ditches onto the ditch bank, but also by disturbing the tall vegetation and creating bare spots providing opportunities for germination and establishment of seeds. The lack of effect of no fertilisation underlines the results we found previously in Chapter 1 and 2 that current measures do not seem to lead to a decrease in the nutrient supply in ditch banks. Of the additional measures, no frequent cultivation (B:0.85, $F_{1,121}$:5.41, P<0.05), no use of pesticides in ditch banks (B:0.73, , $F_{1,121}$:5.64, P<0.05) and extending the zone of no deposition of fertilizer from 1 to 2 meters (B:0.50, $F_{1,121}$:4.26, P<0.05) were linked to the number of target species in ditch banks. This extension to 2 meters has already been incorporated in the prescriptions of the new ditch bank management that was implemented from 2010 on. Figures 1 and 2 may explain why I found no effect of ditch bank management on the number of target species between farmers with and without AES in Chapter 1: There is little difference in the measures implemented between farmers with and without ditch bank management and of the two measures that are performed (no fertilisation has no effect on target species richness, while no deposition of ditch bank slurry has a significant negative effect). Our findings in this synthesis confirm the recommendations of Chapter 1 and 2 that more elaborate measures that aim at decreasing the nutrient supply or competition for light result in higher plant species richness in ditch banks.

5. Interdisciplinary model for ditch bank management

In this thesis I demonstrated what ecological measures should be taken to improve species richness in ditch bank management and I have shown what psychological factors should be addressed to increase farmers' intention to implement these measures. However, combining the fields of ecology and psychology into one interdisciplinary model which shows which psychological variables can be addressed to increase farmers' intention to perform ditch bank management with more mandatory measures and how this will finally result in more target species in ditch banks is still lacking. Based on these previous chapters and the additional data presented in this synthesis I was able to construct this model. To do so I expanded the TPB model we used in Chapter 3 and 4 with the ecological outcome of human behaviour: the number of target species derived from Figure 2.

In this model the performance of only the 2 mandatory measures were used as behavioural variable, because the questionnaire of Chapter 3 inquired specifically about subsidized AES and thus the mandatory measures. In this interdisciplinary model we expected behaviour to be the only variable affecting target species (Fig. 3). I was able to test this model for 127 farmers. I analysed the correlations between the variables in our model and subsequently performed a Structural Equations Model (SEM) using AMOS 21 (IBM, Chicago, IL, USA). I tested for a good model fit by using three goodness of fit indices; the Comparative-Fit-Index (CFI), the Standardized-Root-Mean-Square-Residual (SRMR) and the Root-Mean-Square-Error-of-Approximation (RMSEA). I applied as cut-off values for a satisfactory model fit below 0.08 for RMSEA, 0.08 for SRMR and a CFI of at least 0.90 in line with recommendations from the literature (Hu and Bentler 1999).

Table 1 shows that the number of target species is not correlated with behaviour. Behaviour, though, is correlated with intention, in line with the assumptions of TPB. However, the correlation between self-identity and behaviour is much higher. Our results demonstrate that our model had an acceptable fit (RMSEA: 0.077, CFI: .961, SRMR: 0.057). Attitude was the only variable to have a significant association with intention (Figure 3).

Table 1 Means (M), standard deviations (SD) and correlations of variables in our model for ditch bank management. Significance thresholds **:P<0.01, *:P<0.05

		M	SD	1	2	3	4	5	6	7	8	9
1	Target species	3.58	1.37									
2	Behaviour	0.98	0.54	.015								
3	Intention	4.33	0.98	.151	.255**							
4	Attitude	4.24	0.86	.135	.269**	.539**						
5	Subjective norm	3.2	1.18	.048	.120	.277**	.401**					
6	PBC	3.67	1.32	.222*	.078	.267**	.291**	.003				
7	Self-identity	3.62	1.11	.172	.322**	.414**	.613**	.575**	.242**			
8	Group norm	3.53	1.04	.154	.051	.221*	.375**	.482**	.031	.486**		
9	Group facilitation	3.96	1.00	.058	.218*	.262**	.413**	.443**	.252**	.427**	.451**	
10	Group identity	3.74	0.76	-.109	.009	.005	-.114	-.121	.02	0.054	-.074	-.131

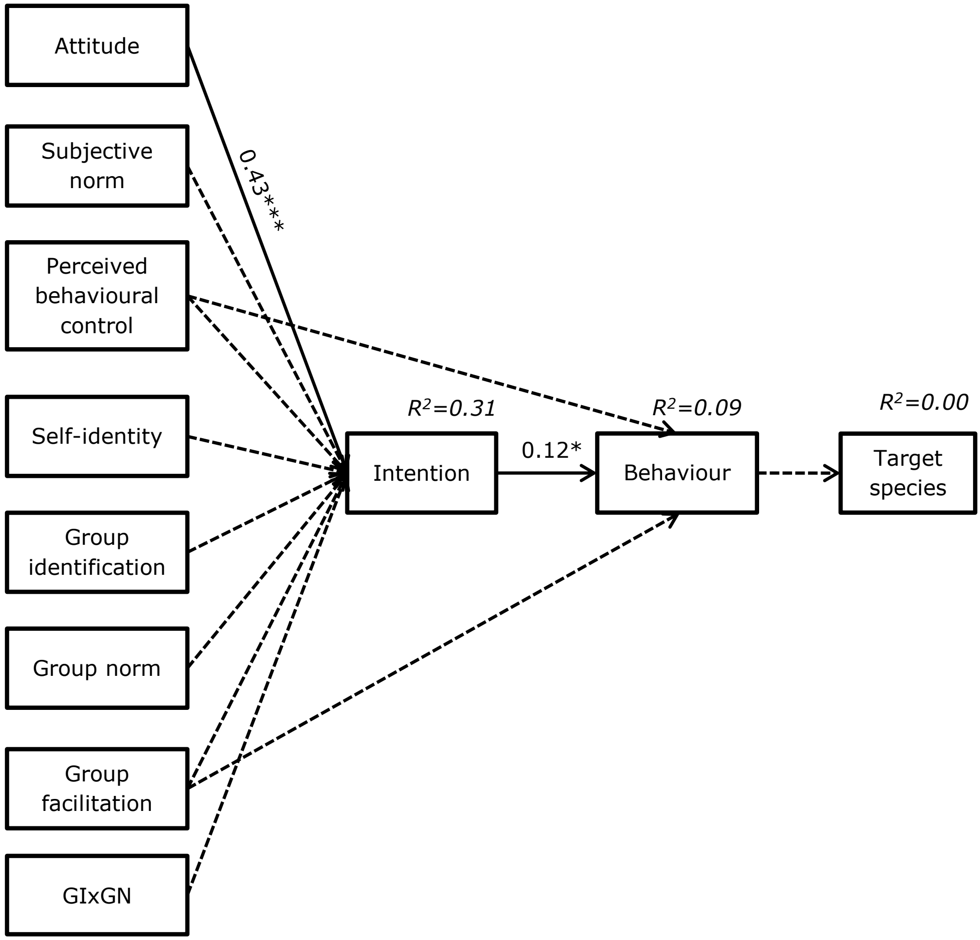

Figure 3 Output of the path analysis from TPB to the number of target species in ditch banks. Only significant betas are shown. The variable behaviour is calculated as the performed number of mandatory measures as prescribed by AES. Values in italics represent the R^2 of the path. GIxGN is the interaction group identification with group norm

Intention was significantly related to the number of mandatory measures. However, I found no significant association between the number of target species and the number of mandatory measures performed, underlining the results of Figures 1 and 2.

6. Where to from here?

I have demonstrated that different variables explain farmers' intention to perform subsidized or unsubsidized agri-environmental measures. In order to change farmers' behaviour effective interventions or governmental campaigns should address the factors that I have shown to be significantly associated with intention. Remarkably, intervening on previously tested relevant psychological variables is, although often advocated, rarely applied in psychological literature (Dwyer et al. 1993, Hardeman et al. 2002, Michie et al.

2008). In the following paragraphs I will further discuss potential interventions specifically aiming at the motivational variables that I found to be associated with the intention to perform subsidized or unsubsidized environmental measures.

6.1. Interventions for subsidized management

To increase farmers' willingness to continue to perform subsidized ditch bank management with more extensive measures I demonstrated that attitude and to some extend perceived behavioural control are the variables to address. The theory of planned behaviour states that underlying to attitude and perceived behavioural control are salient behavioural and control believes about the behaviour (Ajzen 1991). Behavioural beliefs are the beliefs about the likely outcomes of the behaviour and control beliefs are the perceived presence of factors that may facilitate or impede the performance of the behaviour. An intervention technique that addresses both behavioural and control beliefs and has been proven to be effective in establishing a behavioural change is providing feedback (Dwyer et al. 1993, Abrahamse et al. 2005, Michie et al. 2008). Feedback entails providing information about current behaviour and the associated outcomes. This can be done on the individual level or by comparing the outcomes with a group. In the context of AES this approach has been advocated previously by the Snoo by benchmarking the outcomes of AES (de Snoo and Kragten , de Snoo 2006). However, this intervention has shown little effect on behavioural change in farmers so far (Lokhorst et al. 2010). Previous research demonstrated that the effect of providing feedback alone is often limited to short periods of time (Staats et al. 2004). Combining interventions aimed at different variables is assumed to be more successful in achieving a longer lasting change in behaviour (Gardner and Stern 2002).

An alternative intervention technique that can be applied to address both behavioural beliefs and control beliefs is self-monitoring (Michie et al. 2008). Self-monitoring is the recording of the behaviour, or the outcome of the behaviour in for example a diary by the person performing the behaviour. Self-monitoring makes the actual outcomes and control of the behaviour visible and can alter as such the expected outcomes and control beliefs. The positive effect of monitoring the ditch banks by the farmers themselves, was already acknowledged when ditch bank management was initiated in 2000 and was one of the reasons why the payment by results approach was applied (van Strien et al. 1988, Clausman 1996, Musters et al. 2001). In this system farmers' subsidy was reimbursed based on the presence of a number of target species in their ditch banks. The presence of these species was monitored by farmers themselves and subsequently ascertained by the EC. However, from 2009 on under the new Dutch agri-environmental program SNL, payment by result and self-monitoring was abolished. This may have had negative consequences for both farmers' intention to perform ditch bank management and the actual measures performed. Reintroducing self-monitoring may improve farmers' motivation for ditch bank management.

6.2. Interventions for unsubsidized management

In Chapter 4 I demonstrated that for unsubsidized environmental measures self-identity was the variable most significantly associated with intention, followed by attitude, subjective norm, PBC and facilitation by the EC. This indicates that providing feedback and self-monitoring may also alter the intention and performance of additional unsubsidized measures, by addressing attitude and PBC. But to increase the performance of these measures they should be expanded with additional interventions.

A complementary intervention technique that has been demonstrated to be very powerful in changing people's behaviour is commitment making (Burger and Caldwell 2003, Burger and Guadagno 2003, Lokhorst et al. 2010).

The effect of commitment making on behavioural change has been hypothesized to be mediated by self-identity. This pathway is based on self-perception theory. This theory states that people construct their self-image based on their actions rather than the other way around (Bem 1972). So when people perform a certain behaviour by free will, even after committing to it, they will consider themselves as "apparently I am the kind of person who does this", altering their self-identity. In the context of AES, commitment making in combination with feedback has been demonstrated to be effective in increasing the time farmers spend on unsubsidized measures (Lokhorst et al. 2010).

Finally, a third potentially effective intervention method would be labelling. Labelling is the process of making a certain identity more salient (e.g. nature-friendly farmer) by linking that identity to a certain behaviour that a person performs (Cialdini et al. 1998). For example: "You indicated that you are maintaining a hedge on your farm, so you seem like a farmer that cares about nature." This way the behaviour maintaining a hedge is associated with the identity farmer that cares about nature, and this identity is made salient. By subsequently addressing this identity in campaigns for additional measures the farmers may be more willing to perform this behaviour. For example: "You as a nature-friendly farmer may be interested in hanging a nesting box to protect birds". The effect of labelling has been demonstrated to be effective in shifting the behaviour towards behaviour in line with the identity made more salient (Cialdini et al. 1998). In the context of AES this technique has been applied in governmental campaigns for meadow bird management by labelling farmers that perform meadow bird management as "weidevogelboer" (meadow bird farmer), thus making this identity salient based on performed behaviour (meadow bird management). For ditch bank management very little of such campaigns take place. The difference that I found in Chapter 3 between the significance of the association between self-identity and intention for ditch bank management and meadow bird management may be caused by this meadow bird farmer labelling campaign.

7. Future research

In this thesis I made a first step into showing how current ditch bank management can be improved. However, the large scale data approach that I applied requires more in depth field research both from an ecological and a psychological perspective.

From an ecological perspective, more research is required into what measures can be performed to reduce soil nutrient level and competition for light to create suitable circumstances for damp hay meadow species on ditch banks along productive pastures. This research should be complemented by studies into measures enhancing seed dispersal, particularly for dispersal groups currently strongly limited such as species with no capacity to disperse over long distances. Moreover, the feasibility and economic costs of performing environmental measures at productive farmland compared to the effect of nature reserves on productive farmland should be further revised.

From a psychological perspective more in depth research is required into the salient behavioural and control beliefs underlying the TPB variables that I have studied. These beliefs may elucidate which factors are currently believed by farmers to hamper their performance of ditch bank management and could be addressed in future improvements of management or interventions.

This research should be complemented by experimental psychological research into interventions addressing the variables we found to be significantly associated with intention. The effect of these interventions should be measured by performing a pre- and post-interventional TPB questionnaire to measure the change in the TPB variables and the beliefs underlying them. This approach is widely advocated but little applied so far (Hardeman et al. 2002, Michie et al. 2008).

8. Conclusion

In this thesis I conclude that more elaborate measures at locations with a favourable landscape configuration are required to make ditch bank management effective in increasing the plant species richness. I have argued that motivating farmers to extend their management can be implemented by two possible approaches: by changing the prescriptions of subsidized management or by motivating farmers to perform additional unsubsidized measures. I have demonstrated that different motivational variables should be addressed to enhance farmers' willingness to either continue ditch bank management with more elaborate prescriptions or perform additional measures unpaid.

What I have presented in this thesis is an integrated interdisciplinary approach on how to conserve plant species at agricultural ditch banks. By doing so I hope I have highlighted that nature conservation, particularly environmental management at productive farmland is not a one dimensional ecological or social problem, but an interdisciplinary challenge in which both social sciences and ecology are necessary to successfully complement the other.

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English Summary

One of the key policy instruments implemented in Europe to conserve the biodiversity on farmland are Agri-Environment Schemes (AES). However, over the last 10 years various authors have questioned the efficacy of AES in conserving biodiversity in agricultural areas. The improvement of the efficacy of AES in increasing the species richness in agricultural areas has been regarded as both an ecological and social challenge. For a successful implementation of measures with a sound ecological background the willingness of farmers to perform these measures is inevitably needed to have these measures widely implemented. However, until now the two science fields have not been joined into one interdisciplinary study on how to improve AES.

In this thesis I have investigated from both an ecological and psychological perspective what factors hampered the outcomes of AES in the Netherlands over the last 10 years. I have focussed on the management of agricultural ditch banks to restore the biodiversity in these banks. To do so I have used a large scale, long term dataset of the occurrence of target plant species on ditch banks collected by farmers and put them in a landscape ecological perspective to analyse factors affecting species composition. Furthermore, I have performed a questionnaire inquiring about what motivational factors underlie farmers' willingness to participate in subsidized and unsubsidized environmental measure. Finally, I have combined these two research fields into an interdisciplinary model covering the factors underlying farmers' motivation to perform ditch bank management, the measures they perform and how this affects the occurrence of target species in ditch banks.

I demonstrated that over the last 10 years the species richness on ditch banks has slightly increased. However, I found no difference in the increase between ditch banks with and without agri-environment scheme. The small increase in species richness was attributed to an increase of species with a high nutrient demand that disperse by water, while for species that had no capacity to disperse over long distances I found no increase.

Next, I investigated if local nature reserves could act as species-rich sources from which plants can disperse to agricultural ditch banks. I demonstrated that only for water dispersed species there is an increasing colonisation of ditch banks from nature reserves, but this process was limited to a few fast-growing species only.

From a psychological perspective I investigated what factors underlie farmers' willingness to perform subsidized ditch bank management and unsubsidized environmental measures. I found that for subsidized ditch bank management farmers' attitude, that is their expectancies of the outcomes of the management is strongest associated with farmers' intention to perform ditch bank management. However, for unsubsidized environmental measures I found that farmers' perception of whether these measures are associated with their self-identity is the most important determinant of the willingness to perform them.

Our results demonstrate that more elaborate measures at favourable locations in the landscape are required to improve the outcomes of ditch bank management. In order to have this implemented there are two possible approaches: expand current management prescriptions in which the attitude towards management would be the variable to address in interventions or campaigns to enhance willingness among farmers to continue management. Alternatively, self-identity could be addressed to motivate farmers to perform additional unsubsidized measures.

Nederlandse samenvatting

Agrarisch natuurbeheer is een van de belangrijkste beleidsinstrumenten in Europa om de soortenrijkdom in het agrarisch gebied te herstellen. Maar in de afgelopen tien jaar hebben verschillende onderzoeken aangetoond dat tot nu toe de effectiviteit van agrarisch natuurbeheer beperkt is geweest. De verbetering van de effectiviteit van agrarisch natuurbeheer is zowel een ecologische als een sociale uitdaging. Voor een succesvolle implementatie van maatregelen is zowel een stevige ecologische basis als ook de bereidheid van boeren nodig om deze maatregelen op grote schaal uit te voeren. Tot nu toe heeft er echter nog geen onderzoek plaatsgevonden dat de ecologische en maatschappijwetenschappelijke invalshoek met elkaar gecombineerd tot een interdisciplinaire studie die een bijdrage kan leveren aan de verhoging van de effectiviteit van agrarisch natuurbeheer.

In dit proefschrift heb ik vanuit zowel een ecologisch als een psychologisch perspectief onderzocht welke factoren in de afgelopen tien jaar het resultaat van het agrarisch natuurbeheer hebben beperkt. Ik heb me hierbij gericht op het het agrarisch natuurbeheer in slootkanten dat tot doel had om de diversiteit aan plantensoorten langs slootkanten toe te laten nemen. Ik heb hiervoor een grootschalige, lange-termijn dataset over het voorkomen van doelsoorten in slootkanten gebruikt die verzameld is door boeren. Ik heb deze data vervolgens in een landschapecologisch perspectief geplaatst om de factoren die van invloed zijn op de soortensamenstelling te onderzoeken. Daarnaast heb ik een enquête uitgevoerd om te onderzoeken welke motivationele factoren ten grondslag liggen aan de bereidheid van boeren om gesubsidieerde en ongesubsidieerde maatregelen ten gunste van de biodiversiteit uit te voeren.

Tot slot heb ik deze twee onderzoeksbenaderingen gecombineerd in een interdisciplinair model voor slootkantbeheer met daarin de factoren die ten gronslag liggen aan de motivatie van boeren om slootkantbeheer uit voeren, de maatregelen die zij daadwerkelijk uitvoeren en het effect van deze maatregelen op het voorkomen van de doelsoorten in slootkanten.

Mijn resultaten tonen aan dat de soortenrijkdom in slootkanten over de afgelopen tien jaar licht is toegenomen. Ik heb echter geen verschil gevonden in de toename tussen slootkanten met en zonder gesubsidieerd slootkantbeheer. De kleine toename in soortenrijkdom kan worden toegeschreven aan de toename van soorten van voedselrijke milieus die zich via water verspreiden, terwijl soorten die geen aanpassingen hebben om zich over lange afstanden te verspreiden, niet toenamen.

Vervolgens heb ik onderzocht of lokale natuurgebieden kunnen fungeren als soortenrijke bronnen in het landschap vanwaaruit planten zich kunnen verspreiden naar de slootkanten. Alleen voor soorten die zich via het water verspreiden, werd een toename in de kolonisatie van slootkanten gevonden, die beperkt was tot slechts een klein aantal snelgroeiende soorten.

Vanuit een psychologisch perspectief heb ik onderzocht welke factoren ten grondslag liggen aan de bereidheid van boeren om gesubsidieerde en ongesubsideerde maatregelen ten behoeve van de biodiversiteit uit te voeren. Ik vond dat voor gesubsidieerd natuurbeheer attitude, de verwachte uitkomsten van slootkantbeheer, het sterkst geassocieerd was met de intentie om slootkantbeheer uit te voeren. Voor ongesubsidieerde maatregelen vond ik echter dat de perceptie van boeren in hoeverre deze maatregelen correspondeerden met hun zelf-identiteit, de belangrijkste determinant was van de intentie van boeren om deze maatregelen uit te voeren.

Mijn resultaten tonen aan dat er uitgebreidere maatregelen op gunstige locaties in het landschap nodig zijn om de resultaten van slootkantbeheer te verbeteren. Om dit in de praktijk te implementeren zijn er twee mogelijke benaderingen: het uitbreiden van de huidige maatregelen waarbij de attitude ten op zichte van het beheer de psychologische variabele is die geadresseerd wordt in interventies of overheidscampagnes. De tweede mogelijkheid is om de zelf-identiteit te adresseren om boeren te motiveren om extra ongesubsidieerde maatregelen uit te voeren.

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I was wondering if I was a suitable candidate, given my lack of interest in social sciences. But I still went for the interview. I made it for the job and started to develop an interest in social sciences. Now after 4 years there is a thesis that contains two ecological chapters, two social psychological chapters and an interdisciplinary synthesis. And I must say that I've worked over the last 4 years with lots of fun on this project. What made it so interesting for me was that it applied a challenging interdisciplinary approach in a politically precarious branch of nature conservation. However, thanks to the contributions, both great and small from a large number of people, the experience of the PhD would not have been the same for me.

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Curriculum vitae

William van Dijk was born on 31 December 1984 in Zevenaar, the Netherlands. After graduating from Liemers College in the year 1997 he started studying biology at Radboud University Nijmegen. During his MSc he did an internship at the department of Experimental Plant Biology at the Radboud University on the timing of induced resistance to herbivory in clonal networks. He did his second internship at the department of Plant Ecology and Biodiversity at Utrecht University in collaboration with the Instituto Boliviano de Investigación Forestal. He investigated population dynamics of tree species using population matrix models for sustainable timber harvesting in Bolivian dry forests. He did his third internship at the Radboud University Nijmegen on the relationships in plant composition between Dutch floral districts based on European distribution patterns of these plants. After obtaining his MSc degree in Biology in August 2009 he started the educative master to become a high school teacher at the Radboud University Nijmegen and Liemers College. In February 2010 he started as a PhD candidate at the department of Plant Ecology and Nature Conservation at Wageningen University. In his project he studied the ecological and psychological factors underlying the outcomes of Dutch agri-environment schemes. Furthermore in 2011 he became a member and in 2013 the chair of the PhD Council of the C.T. de Wit Graduate School for Production Ecology & Resource Conservation (PE&RC) and held a position in the Board of PE&RC. From 2011 on he became also a member of the Wageningen University PhD Council and represented Wageningen University PhD candidates in PhD Network Netherlands (PNN). From Februari 2014 on he started a Post-doctoral position at the University of Aberdeen on genome-wide association mapping and landscape scale modelling of heritable ionomic diversity in Arabidopsis thaliana populations.

List of publications

van Dijk, W.F.A., Van Ruijven, J., Berendse, F., de Snoo, G.R., 2014. The effectiveness of ditch banks as dispersal corridor for plants in agricultural landscapes depends on species' dispersal traits. Biological Conservation 171, 91-98

van Dijk, W.F.A., Schaffers, A.P., Leewis, L., Berendse, F., de Snoo, G.R., 2013. Temporal effects of agri-environment schemes on ditch bank plant species. Basic and Applied Ecology 14, 289-297.

Van Dijk, W.F.A., Schaffers, A.P., Van Ruijven, J., Berendse, F., De Snoo, G.R., 2013. Shifts in functional plant groups in ditch banks under agri-environment schemes and in nature reserves. Aspects of Applied Biology 118, 71-79.

Gomez S., van Dijk W.F.A. & Stuefer J. (2010) Timing of induced resistance in a clonal network. Plant Biology 12 (3): 512 - 517.

van Dijk, W.F.A., Lokhorst,A.M., Berendse, F., de Snoo, G.R., (in review) Collective agri-environment schemes: How can regional environmental cooperatives enhance farmers' intentions for agri-environment schemes? Land use policy

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PE&RC Training and Education Statement

With the training and education activities listed below the PhD candidate has complied with the requirements set by the C.T. de Wit Graduate School for Production Ecology and Resource Conservation (PE&RC) which comprises of a minimum total of 32 ECTS (= 22 weeks of activities)

The C.T. De Wit Graduate School PE&RC PRODUCTION ECOLOGY & RESOURCE CONSERVATION

Review of literature (4.5 ECTS)
- Benchmarking biodiversity of farmers

Writing of project proposal (4.5 ECTS)
- Benchmarking biodiversity of farmers

Post-graduate courses (3.9 ECTS)
- Mixed linear models; PE&RC (2011)
- Introduction to R for statistical analysis; PE&RC (2011)
- Generalized linear models; PE&RC (2011)
- Bayesian statistics; PE&RC (2013)
- Spatial ecology; PE&RC (2013)

Invited review of (unpublished) journal manuscript (2 ECTS)
- Biological Conservation: predicting dispersal-limitation in plants: implications for restoration of isolated wetlands in agricultural landscapes (2012)
- Agriculture, Ecosystems and Environment: contribution of woody habitat islands to conservation of birds and their ecological functions in an extensive Colombian rangeland (2013)

Deficiency, refresh, brush-up courses (0.9 ECTS)
- Linear models (2010)

Competence strengthening / skills courses (3 ECTS)
- PhD Competence assessment; WGS (2010)
- Social psychology; Coursera (2013)

PE&RC Annual meetings, seminars and the PE&RC weekend (2.4 ECTS)
- PE&RC Weekend; first year (2010)
- PE&RC Day : selling science (2010)
- PE&RC Day: what ate the neighbours doing (2011)
- PE&RC Weekend; final year (2012)
- PE&RC Day: extreme life (2012)
- PE&RC Day: biomimicry (2013)

Discussion groups / local seminars / other scientific meetings (7.5 ECTS)

- PE&RC-FORCON: forest and conservation ecology (2010-2013)
- Wageningen Ecology & Evolution Seminars (2010-2013)
- CBS-Minisymposium orde uit de chaos (2013)
- Startsymposium leerstoel agrarische natuurbeheer (2013)
- Slotsymposium leerstoel agrarische natuurbeheer (2013)
- Sustainable intensification of Agricultural Systems (2013)

International symposia, workshops and conferences (6 ECTS)

- NERN-Meeting; poster presentation (2010)
- European Congress of Conservation Biology; oral presentation; Glasgow, UK (2011)
- Environmental Management on Farmland; oral presentation; Brigg, UK (2013)
- NERN-Meeting; oral presentation (2013)

Lecturing / supervision of practical's / tutorials (3 ECTS)

Ecologie (2011, 2012)
Veldpracticum Bos- en natuurbeheer (2012-2013)
Verdiepende Casus omgevingswetenschappen (2013)

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