

Temporal Effects of Organic Farming on Biodiversity and Ecosystem Services

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Cover: Organic cereal field infested with *Sonchus arvensis*
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Abstract

Agricultural intensification has caused a dramatic decline of global biodiversity and associated ecosystem services. Organic farming has been shown to partially counteract agricultural intensification by applying environmentally friendly and resource efficient farming practices, but opportunities to improve in efficiency still remain. This thesis investigates the contribution of organic farming to biodiversity and ecosystem services with focus on the effect of the time since transition (TST) to organic farming methods.

Surveys on butterflies, plants, moths, carabid beetles and an experimental study on weed seed predation were performed on conventional and organic farms situated in landscapes differing in landscape complexity. The organic farms had been under organic management between 1 and 25 years before surveys. This design allowed for analyses of the effect of organic farming while accounting for the time since transition and landscape composition.

The overall effect of organic farming was small. Only butterflies and plants (in one out of two studies) had higher species richness and abundance on organic compared to conventional farms. However, analyses of the time since transition to organic farming revealed novel facts: butterfly abundance increased gradually by 100% over 25 years, whereas butterfly and plant species richness increased rapidly at the transition and then remained fairly constant. The moths that initially did not appear to increase in the organic farming system showed a clear positive response to newly transitioned farms ($TST \leq 6$ years), whereas conventional and old organic farms ($TST \geq 15$ years) had similar diversity. Two plant species occurred more frequently on new organic farms and two species on old organic farms. Neither carabids nor seed predation showed any temporal responses to organic farming.

This thesis shows that explicitly addressing temporal effects of organic farming may result in novel and unexpected findings. Control for temporal effects opens up for better understanding of the complexities between organic farming, biodiversity and ecosystem services over time. Future evaluations need to address this factor for high credibility and usefulness in the development of improved policies for organic farming.

Keywords: Agri-environment schemes, agricultural intensification, arable weeds, biological control, Carabidae, farmland biodiversity, farmland conservation, landscape complexity, Lepidoptera, traits.

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List of Publications

This thesis is based on the work contained in the following papers, referred to by Roman numerals in the text:

- I **Jonason, D.**, Andersson, G. K. S., Öckinger, E., Rundlöf, M., Smith, H. G., Bengtsson, J. (2011). Assessing the effect of the time since transition to organic farming on plants and butterflies. *Journal of Applied Ecology*, 48(3), 543-550.
- II **Jonason, D.**, Andersson, G. K. S., Öckinger, E., Smith, H. G., Bengtsson, J. (2012). Field scale organic farming does not counteract landscape effects on butterfly trait composition. *Agriculture, Ecosystems and Environment*, 158, 66-71.
- III **Jonason, D.**, Franzén, M., Pettersson, L. B. Transient peak in moth diversity as a response to organic farming. (manuscript)
- IV **Jonason, D.**, Smith, H. G., Bengtsson, J., Birkhofer, K. Landscape simplification promotes weed seed predation by carabid beetles (Coleoptera; Carabidae). (manuscript)

Paper I and II are reproduced with the permission of the publisher.

The contribution of Dennis Jonason to the papers included in this thesis was as follows:

- I Main author and analysis. Study idea and design with Jan Bengtsson, Erik Öckinger, Henrik Smith, Georg Andersson, and Maj Rundlöf. Field work with Georg Andersson.
- II Main author and analysis. Study idea and design with Erik Öckinger and Jan Bengtsson. Field work with Georg Andersson.
- III Main author, study idea, field work and analyses. Study design with Lars Pettersson.
- IV Main author and study idea. Design with Klaus Birkhofer and Henrik Smith. Field work with Klaus Birkhofer.

1 Introduction

1.1 Agricultural intensification

In line with the increased human demand for food and the post-war technical advancement, most countries have undergone major agricultural intensification. Old farming traditions using more extensive farming methods have been abandoned in favour of specialised farms with large input of synthetic fertilisers and pesticides (Foley *et al.*, 2005; Stoate *et al.*, 2009). On one hand this has resulted in an unprecedented increase in agricultural productivity, but on the other hand it has been criticized for being the foremost cause behind a worldwide biodiversity decline (Kleijn *et al.*, 2009). Pesticide use has the most consistent negative effects on biodiversity (Geiger *et al.*, 2010), but factors such as landscape homogenisation and fragmentation through the conversion of native and semi-natural ecosystems to agriculture also contribute (Benton *et al.*, 2003; van Swaay *et al.*, 2006). It is the reliance on external inputs and the process of landscape homogenisation that I henceforth will refer to as agricultural intensification.

1.1.1 Biodiversity and landscape composition

The intensification of agriculture has resulted in drastic land use changes in Sweden, and elsewhere, where wetlands, meadows and other natural or semi-natural habitats largely have disappeared in order to increase the land available for cultivation (Ihse, 1995; Tschardtke *et al.*, 2005). For biodiversity the consequences have been severe with huge habitat losses, subsequent habitat fragmentation, and increased isolation of remaining habitats.

Biodiversity in homogeneous arable landscapes is largely confined to the remaining fragments of semi-natural habitats, e.g. grasslands (Öckinger & Smith, 2007). As these habitats usually are both small and isolated, the inhabitant species become more sensitive to stochastic extinctions. The species may, however, persist in the landscape with help from individuals immigrating

from habitat patches of better quality, so called source habitats (Pulliam, 1988; Hanski, 1999). Consequently, large and well-connected semi-natural habitats are fundamental for biodiversity in landscapes dominated by agriculture (Hanski & Ovaskainen, 2000; Öckinger & Smith, 2007), and its reduction adds to the problem of landscape homogenisation and biodiversity losses.

1.1.2 Species traits

Not all species are equally affected by land use and environmental changes; species with certain traits may be more resilient than others (Swift *et al.*, 2004). In the context of agricultural intensification and landscape homogenisation, one can make following predictions: (i) mobile species are better able to exploit spatially separated resources compared to sedentary species (Hanski & Ovaskainen, 2000) and are also more likely to re-colonise faster after habitat improvement in largely homogeneous landscapes where distinct high-quality habitat patches constitute population sources (Öckinger & Smith, 2007); (ii) resource generalists, compared to specialists, have the opportunity to utilize alternative resources (like non-preferred host plants) if patches of the preferred resource are isolated from each other. Hence, generalist species will be more common in intensively managed homogeneous landscapes (Warren *et al.*, 2001); (iii) species with high reproductive rates can recover more rapidly from population declines and require lower amounts of habitat for population persistence in homogeneous landscapes, compared to species with lower reproductive rate (Vance *et al.*, 2003; Fahrig *et al.*, 2011). Based on these predictions, high biodiversity may not necessarily imply high trait diversity if a type of disturbance affects species with different traits disproportionately.

The study of traits-related community composition in relation to anthropogenic activities has lately received much scientific interest (Webb *et al.*, 2010). Trait diversity, as opposed to species diversity, allows for detection of patterns in species responses and predictions concerning other taxa with shared ecological characteristics (Verheyen *et al.*, 2003), and may reveal additional information which otherwise would remain hidden. Trait diversity can also be used as a predictor of ecosystem processes and functions as species with different traits perform different functions in the ecosystems (Vandermeer *et al.*, 1998; Diaz & Cabido, 2001). Thus, when species go extinct and communities become structurally homogenised, this may alter the functioning, productivity and resilience of ecosystems, as well as the services society obtain from them (Olden *et al.*, 2004; Clavel *et al.*, 2010).

Most studies of diversity responses to management practices or landscape structure are still performed on species richness, despite the supplementary information that may be obtained by analyzing species traits. A probable

explanation to this is that species diversity is much easier to measure than trait diversity as it sometimes can be difficult to distinguish which species that holds a specific trait or which trait that relates to a certain function. Further, species that at one point in time seem to be redundant may hold traits important at times of environment change, which emphasizes the importance of high biodiversity as insurance for ecosystem functioning (Loreau *et al.*, 2001).

1.2 Organic farming

Within the European Union, €34.4bn has been assigned to agri-environment schemes (AES) between 2007 and 2013; a substantial part of the EU budget (Farmer *et al.*, 2008). AES compensate farmers for income losses and expenditures associated with measures taken to reduce the strain on biodiversity and environment within farmland (Kleijn *et al.*, 2011). Typical measures include restoration or creation of non-crop habitats, creation of winter stubbles and spray free zones (Vickery *et al.*, 2002).

Organic farming is mainly part of AES thanks to the exclusion of synthetic fertilizers and pesticides, which are two key factors separating organic from conventional farming practices (Petersen *et al.*, 2006; Geiger *et al.*, 2010). The regulations of organic farming within the EU follow Commission Regulation No 889/2008 (EC, 2008) and its amendments, but certification organisations, such as the Swedish organisation KRAV, can have supplementary rules. In short, organic farming can be summarised as:

...an overall system of farm management and food production that combines best environmental practices, a high level of biodiversity, the preservation of natural resources, the application of high animal welfare standards and a production method in line with the preference of certain consumers for products produced using natural substances and processes. The organic production method thus plays a dual societal role, where it on the one hand provides for a specific market responding to a consumer demand for organic products, and on the other hand delivers public goods contributing to the protection of the environment and animal welfare, as well as to rural development (EC, 2007).

1.2.1 Organic farming and biodiversity

Organic farming involves alternative methods that in the absence of synthetic fertilizers and pesticides aim to increase soil fertility and the resistance against pests. Soil fertility is maintained through application of animal manure and through nitrogen fixing leguminous plants, whereas pest and weeds are

controlled using varied crop rotations with inclusion of lays and through mechanical weed control. These methods contribute to an overall lower farming intensity on organic farms.

Most studies that have assessed the effect of organic farming, relative to conventional farming, have reported positive effects on biodiversity (Bengtsson *et al.*, 2005; Hole *et al.*, 2005). Still, the relationship between organic farming and biodiversity is not straightforward. Confounding factors may co-vary either with the farming system or with the species' responses. For example, ecological theory predicts highest effect of organic farming at intermediate levels of landscape heterogeneity (Tschamtkke *et al.*, 2005). This theory assumes that a minimum amount of non-crop habitat and connectivity is needed in the landscape to generate source populations for organic land (see also section 1.1.1). In contrast, the effect of organic farming will approach zero in the most homogeneous landscapes due to lack of source habitats and in the most heterogeneous landscapes due to an already saturated biodiversity (Concepción *et al.*, 2012). That many studies have found highest effect in homogeneous landscapes (Rundlöf & Smith, 2006; Holzschuh *et al.*, 2007; Batáry *et al.*, 2011) may result from a concentration effect where species tend to aggregate at the few resources available, resulting in a imaginary large effectiveness of organic farming in homogeneous compared to heterogeneous landscapes (Kleijn *et al.*, 2011).

One confounding factor that may have led to a misinterpretation of the effect of organic farming on biodiversity is time. Several studies have hypothesised that there may be a time-lag in species responses to the benefits of organic farming and consequently that biodiversity will increase with increasing time since transition (Younie & Armstrong, 1995; Hyvönen, 2007; Andersson *et al.*, 2010). This has, however, hardly received any consideration in the literature. If not confounding factors that may impinge upon the effect of organic farming are controlled for, there is a risk that the result received from evaluations to a larger extent derives from these factors rather than the actual factor of interest, i.e. organic farming.

1.2.2 Effects of the time since transition to organic farming

The responses of biodiversity to land use and environmental changes can be immediate, but do usually range over longer time frames (Chamberlain *et al.*, 2000; Kuussaari *et al.*, 2009; Jackson & Sax, 2010). This applies both to negative and positive changes. For example, if the quality of a patch (or landscape) becomes degraded, species may suffer from an extinction debt (Tilman *et al.*, 1994) where the threshold condition for further persistence is no longer met, although the species are still present due to a time lag in the

response to the environmental change. “Living dead” populations resulting from altered and intensified land use has been found amongst several organism groups (Lindborg & Eriksson, 2004; Kuussaari *et al.*, 2009; Sang *et al.*, 2010). On the other hand, if the quality of a patch improves, which is predicted to occur after the transition from conventional to organic farming, the patch may instead experience a colonization credit. This credit constitutes the difference between the number of species present in a patch and the theoretical richness based on the improved patch quality (Cristofoli & Mahy, 2010). The size of the colonization credit can depend on numerous factors such as habitat connectivity, proximity to source areas for dispersing species, vegetation succession, species interrelations and species traits (Jackson & Sax, 2010).

As mentioned previously, possible time-lags in species responses to organic farming have largely been overlooked. Organic farming, as well as other agri-environment schemes, are often applied to areas without regards to any target species or to areas where the target species is missing, with the intent of improving conditions necessary for (re)colonization (Whittingham, 2007). During such circumstances one would expect a time-lag in the species’ response determined by, for example, a combination of species traits and landscape context. Hence, a prolonged response time is expected in homogeneous landscapes due to few source habitats and high isolation (Jackson & Sax, 2010; Perfecto & Vandermeer, 2010), unless the species in question has high dispersal ability. Conversely, the time lag is expected to be shorter in heterogeneous landscapes as a result of more habitat patches and shorter colonization distances.

At what temporal scale species respond to organic farming may also depend on the specific farming practices. Compared to conventional farming in which weeds are controlled directly using herbicides, organic farming utilizes indirect methods, e.g. diversified crop rotations. The indirect methods are slower in their effect (Bàrberi, 2002), which may result in a transition period with higher weed levels during the first years of organic management, followed by a decrease. Weeds comprise a key resource in intensive agricultural landscapes, and a transition period in weed diversity may therefore be mirrored by several weed dependent species.

As a way of recognizing the importance of temporal effects in evaluations of organic farming and other AES on biodiversity, authors occasionally state for how long their farms have been under organic management (e.g. Wickramasinghe *et al.*, 2004; Boutin *et al.*, 2011), but few have controlled for or analyzed temporal effects explicitly. This may obscure important information needed for a comprehensive interpretation of the results. This thesis adds important information to the understanding of the temporal effects

related to the transition between farming systems, by using a design that carefully selected farms differing in time since transition to organic farming and at the same time controlled for potentially confounding effects of landscape composition.

1.3 Ecosystem services

Ecosystem services are defined as the benefits people obtain from ecosystems (Millennium Ecosystem Assessment, 2005). We are sometimes aware of the ecosystem services, for example when a bumblebee pollinates an apple blossom and we later can harvest an apple, but mostly they are simply taken for granted. Other examples of ecosystem services are water supply and purification, climate regulation, nutrient cycling, recreation, and as in this thesis, biological control.

Anthropogenic activities are sometimes altering the dynamics of ecosystems to the extent that it jeopardizes their capacity to function and to buffer environmental disturbances (Elmqvist *et al.*, 2003). On a global scale, 15 of 24 (63%) ecosystem services examined in the Millennium Ecosystem Assessment (2005) are utilized in an unsustainable manner. Changes in land use, for example as a result of agricultural intensification, are indisputably one of the major contributors to the threats of ecosystem services seen today (Foley *et al.*, 2005; Tscharntke *et al.*, 2005). Although increased ability to take advantage of what the ecosystems have to offer has increased human well-being, overuse likely undermine the capacity of ecosystem to function and deliver essential services in the long run (Foley *et al.*, 2005; Millennium Ecosystem Assessment, 2005; Rockström *et al.*, 2009).

The ecosystem services provided by biodiversity can result in significant environmental and economic benefits. Östman *et al.* (2003) found that on insecticide free fields the increase in yield attributed to natural enemies of aphids corresponded to the yield increase attributed to the use of pesticides. Promoting biodiversity and associated ecosystem services may therefore be an environmentally friendly alternative to the use of synthetic agrochemicals that add little or no risk to productivity or farm economy. The challenges for the future lie in managing healthy ecosystems sustainably and in reversing the degradation of ecosystems at risk while meeting the societal demands. This will require a deeper understanding about the relationship between biodiversity and ecosystem functioning and the interplay with land use management at local and landscape scales (Loreau *et al.*, 2001; Tscharntke *et al.*, 2005).

2 Thesis aims

The general aim of this thesis is to disentangle the contribution of temporal effects in evaluations of organic farming on biodiversity and ecosystem services, and to aid the development of novel policy measures aiming at sustainable agriculture and farmland conservation.

More specifically, the questions addressed are:

- Can organic farming benefit biodiversity and the delivery of ecosystem services?
- Do species responses to organic farming and the delivery of ecosystem services change with the time since transition?
- Do the effect of organic farming and the time since transition vary depending on species traits or landscape context?

3 Methods

3.1 Farm selection

The farm selection procedure was essentially the same in all papers. Farms were selected in both of the regions of Uppland and Scania (Figure 1) except from in paper III, which was solely conducted in Uppland. 18, 36, or 60 farms were used depending on paper, of which one third was managed conventionally and two thirds organically. The organic farms had all previously been managed conventionally and were selected based on their time since transition (TST) from conventional to organic farming, which ranged between 1 to 25 years. In paper I and II the gradient in TST was continuous whereas in paper III and IV it categorized farms as new ($TST \leq 6$ years) and old ($TST \geq 15$ years) organic, respectively.

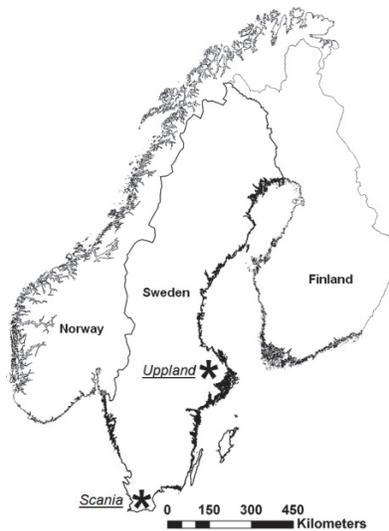


Figure 1. Geographical location of the study regions Uppland and Scania, Sweden.

The effect of organic farming may interact with the composition of the landscape (Tschardt *et al.*, 2005; Concepción *et al.*, 2012; Winqvist *et al.*, 2012). This was accounted for by selecting farms along a gradient of landscape heterogeneity, such that landscapes ranging from low to high levels of heterogeneity should be represented by both conventional farms and by organic farms differing in TST (Figure 2). Landscape heterogeneity was measured as the proportion of arable land within a 1 km radius from each farm and was later corrected to be based from each study field. This measure is commonly used in the literature due to its negative correlation with e.g. the Shannon index of habitat diversity and the proportion of grassland (e.g. Gabriel *et al.*, 2006; Holzschuh *et al.*, 2007).

On each farm, one to two fields were selected for data collection. The crop type was standardized to cereal which allowed for control of possible effects of crop type. In paper III crop type was not standardized.

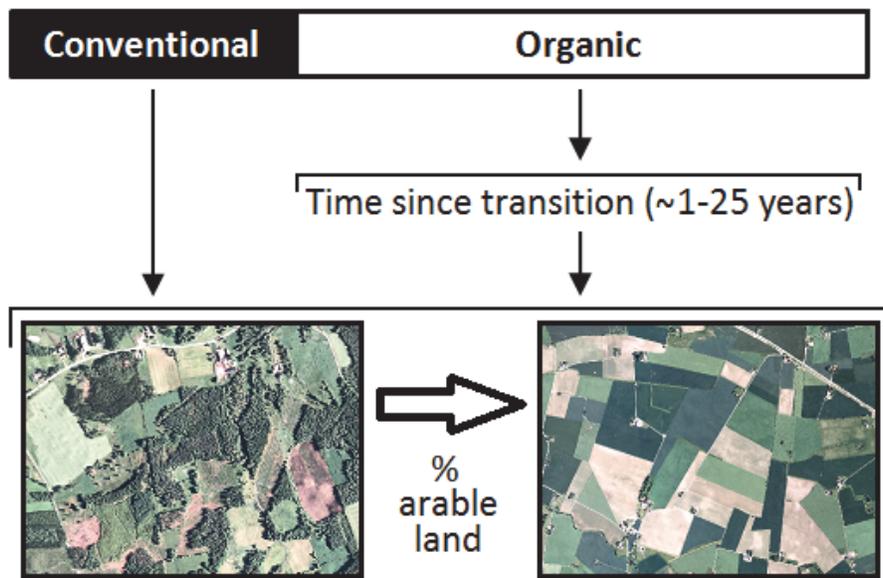


Figure 2. Illustration of the farm selection design. One third of the farms in each study were conventional and two thirds were organic. The organic farms differed in time since transition between 1 and 25 years. All farms were located in landscapes differing in heterogeneity (i.e. proportion of arable land within 1 km) such that farms, irrespective of farming system and time since transition, should be represented in all landscape types (photo: © Lantmäteriet).

3.2 Data collection

3.2.1 Papers I and II, butterflies and plants

Butterflies (Rhopalocera and burnet moths, Zygaenidae) were surveyed on a total of 60 farms divided equally between Uppland and Scania. The time since transition varied between 1 and 25 years. On each farm, a 250 m long transect was set up in the preexisting uncropped margin to a cereal field. The opposite side of the field margins only comprised cereal fields of the same farming system, small gravel roads or small ditches, to as far as possible eliminate factors that may impinge upon what to find in the transect. Two 50 m long transects were also set up within the fields, perpendicular to the margin. The surveys were performed between June and August, 2009, at five (Uppland) and six (Scania) occasions using a standardized method where all individuals 5 m ahead, 5 m into the field and 1.5 m into the margin were identified to species (Pollard, 1977). Surveys were conducted during daytime (9 AM - 5 PM), in sunny conditions at temperatures of 17 °C and above, and at low wind levels (≤ 4 on the Beaufort scale). Species may vary in their diurnal activity pattern, i.e. some are more active in the morning and some in the afternoon, which likely affect their detectability (Wikström *et al.*, 2009). The surveys on each farm were therefore performed at different times of the day.

Herbaceous plants (grasses included) were surveyed twice, at the end of June and July, in the same transects as the butterflies. Species were recorded as presence/absence using 10 inventory squares (30x30 cm) that were evenly allocated along the field margin *c.* 25 cm from the crop edge. Within fields, five inventory squares per transect were placed at a distance of 1, 5, 10, 20 and 40 m from the crop edge.

In paper II, all analyses were made on the data collected for paper I, but here with a focus on butterfly traits. Hence, no additional field data were required. Data on butterfly traits data were collected from the literature (Bink, 1992; Eliasson *et al.*, 2005).

3.2.2 Paper III, moths and plants

Data on moths and herbaceous plants were collected on 18 farms in Uppland categorized as conventional, new organic (TST ≤ 6 years) and old organic (TST ≥ 15 years), with six farms in each category. The farms were located in both heterogeneous and homogeneous landscapes (25-97% arable land).

Moth surveys are mainly performed using either of two methods; light trapping or bait trapping. Light trapping build on the tendency of moths to fly towards lights whereas bait trapping uses olfactory cues (Pettersson & Franzén, 2008). In comparable European studies assessing the effect of organic farming

or other types of agri-environment schemes on moth diversity, light traps have exclusively been used (ISI Web of Science® search words: moth + bait + AES/organic farming). However, at high latitudes as in Sweden, the trapping efficiency of light traps decreases due to the bright summer nights and bait traps were therefore chosen for this study. Bait traps are also less labor-intensive and attract species during both day and night-time, although they only perform in the latter part of the season when there is a selection for olfactory cues. The species attracted to the two trap alternatives may differ in composition (Söderman, 1994), but as the aim with the study solely was to compare catches between farm and landscape types, comparisons between sampling methods become of minor importance.

Jalas bait traps (Figure 3) were tightly secured on poles located 50 m apart in the uncropped margin to one field per farm ($n_{\text{trap}} = 3$ per farm). In this study no standardization for crop type was made. The traps were baited with a saturated mixture of red wine and white sugar soaked into a cloth from which the moths could feed. The cloth was attached to a container with the bait solution to assure that it would not dry out. Attracted moths fell into the lower compartment of the trap where they were slowly killed by chloroform evaporating from a chloroform container. To avoid damaging the collected specimens, the lower compartment contained egg trays between which the moths could hide. The traps were out in field during four consecutive days in the beginning of August, 2010. After emptying of the traps the moths were frozen until identification.

At the same time and place as the moth trapping was performed, the species richness and frequency of herbaceous plants (grasses included) were surveyed using eight plant inventory squares (50x50 cm) divided into 25 segments (10x10 cm). The inventory squares were randomly placed along the margin approximately 25 cm from the crop edge. The number of inventory squares and segments with plant occurrence measured species richness and frequency, respectively.

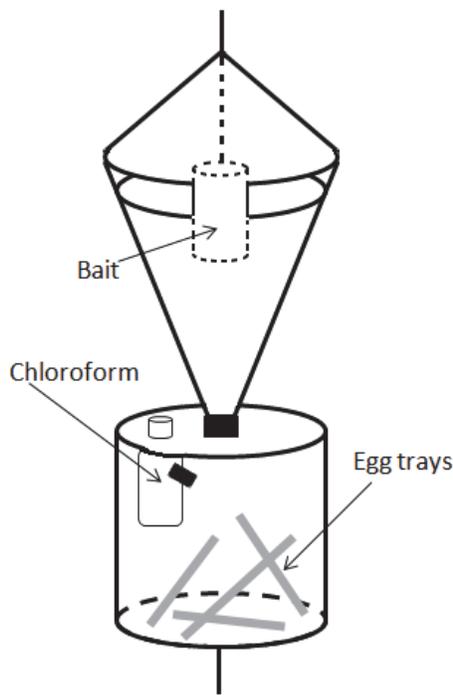


Figure 3. Illustration of a Japas bait trap. The moths attract to the bait consisting of sugar saturated red wine. They subsequently fall into the lower compartment where they hide between egg trays before they slowly get killed by the chloroform.

3.2.3 Paper IV, Carabids and weed seed predation

Carabids (Coleoptera: Carabidae), or ground beetles, are well adapted to disturbances and can be found in large numbers in arable fields where they are preferred for their biological control services (although some species are pest themselves) (Thiele, 1977; Östman et al., 2001). Not all carabids eat seeds on a regular basis, but most of them have wide food ranges which together with their relatively high diversity in arable land make them suitable model organisms in seed predation studies (Menalled et al., 2007; Lundgren, 2009; Fischer et al., 2011).

Weed seed predation in arable land is commonly quantified by placing a known number of seeds within fields and after a certain period of time counting the number of seeds remaining. This method is similar to those used to quantify other types of biological control, for example aphid predation (e.g. Östman *et al.*, 2001). Disappeared seeds are assumed to have been predated on, but they might as well only have been removed and cached for later consumption or dropped along the way, which would not necessarily impair their ability to germinate at a later stage (Vander Wall *et al.*, 2005). Although the actual fate of the seeds was not explicitly examined, the number of removed seeds was used as proxy for the service of weed seed predation (*sensu* Westerman *et al.*, 2003; Fischer *et al.*, 2011).

25 seeds from each of the weed species *Viola arvensis*, *Stellaria media* and *Capsella bursa-pastoris* were lumped together in plastic cups (Ø 75, height 35 mm; Figure 4) and placed in two fields per farm according to the design of Figure 5. If only one suitable field per farm could be find, the groups of cups were placed within this field but at a distance of >100 m from each other. The cups had four 2 cm openings cut out in the side to allow ground dwelling seed eaters access to the seeds. The seeds differed in weight (*V. arvensis* = 0.62; *S. media* = 0.48; *C. bursa-pastoris* = 0.12 mg) to account for different seed size preferences among the seed eaters. Roofs to protect from rainfall and seed predation by birds were made out of cardboard and placed approximately 10 cm above the cups. The seeds were exposed to seed predators in the fields twice during four consecutive days, once in the middle and once by the end of July 2011.



Figure 4. A seed cup in field. (Photo: D. Jonason)

Carabids were sampled using six pitfall traps (Ø 90 mm) per farm filled to one third with 50% propylene glycol as preservative and to reduce surface tension. The traps were open during the periods that the seeds were exposed to seed predators and sealed by lids between sampling periods to prevent unintended catches. The catches were preserved in 70% ethanol until identification.

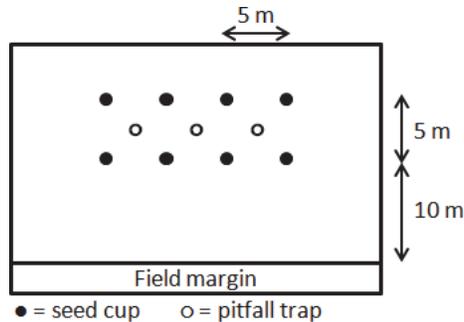


Figure 5. The spatial set-up of seed cups and pitfall traps within field. The set-up was replicated twice per farm; in two fields or separated by >100 m within one field.

3.3 Statistical analyses

The first two papers used a similar statistical approach where a response variable (e.g. species richness) was tested against a combination of predictor variables and their interactions (e.g. farming system and time since transition) using generalized linear models. This approach results in a number of model combinations where each predictor variable is included in several models. The relative strength of each individual model, given the parameters, was assessed based on Akaike's Information Criterion (AIC) corrected for sample size ($=AIC_c$) (Akaike, 1974). AIC can recognize models with similar fit and assigns each model a weight, i.e. a probability value of having best fit among all analyzed models (Whittingham *et al.*, 2006). As no single best model was found (i.e. $\Delta AIC_c < 2$; Burnham & Anderson, 2002), models were averaged to calculate average parameter estimates for each predictor variable where the contribution of each model combination was proportional to its weight. In paper II farm identity was included as a random factor (= mixed models).

Paper III tested the effects of farming system, farm category (i.e. conventional, new organic and old organic) and landscape composition on moths and plants using analysis of covariance (ANCOVA), which also is a type of linear model. Differences in moth community composition between farm categories were tested using permutational multivariate analysis of variances (PerMANOVA). Single species associations to the farming systems and to the farm categories were tested using indicator species analysis (ISA).

PerMANOVAs were also applied in paper IV, this time to test for differences in carabid diversity and seed predation among farming systems, farm categories, landscape types and regions (Uppland and Scania).

4 Results and discussion

4.1 Effects of organic farming

4.1.1 Biodiversity

All papers studied the effect of organic farming on biodiversity, but no consistent result could be found. The species richness, abundance or frequency were higher for all species on organic compared to conventional farms, but the variation between farms was high and although the differences were seemingly large, not all were statistically significant (Table 1). This confirms the general findings of high variability in the effect of organic farming on biodiversity (Bengtsson *et al.*, 2005; Hole *et al.*, 2005; Winqvist *et al.*, 2012).

The most pronounced effect of organic farming was found on butterflies, having on average 20% higher species richness and 60% higher abundance compared to conventional farms (paper I). These results can probably be explained by the higher plant species richness on organic farms (20%), as plants constitute an important resource for butterflies in terms of nectar etc. The higher plant species richness contrasts the results of paper III where no effect of organic farming on either plant species richness or frequency was found. However, when the data in paper III were analyzed at the species level, one species (*Cirsium arvense*) seemed to benefit from organic farming (see also section 4.2 for temporal effect of organic farming). This shows that the effect of organic farming not only varies between, but also within species groups.

No difference in moth species richness or abundance was found between farming systems (paper III; Table 1, but see section 4.2). If it holds true that plants drive the responses of other species depending on them (e.g. Steffan-Dewenter & Tscharntke, 2001), then the non-significant effect of organic farming on the plants may explain the lack of response among the moths.

Table 1. Diversity measures and percentage seed removal on organic and conventional farms. Average \pm SE and relative difference (%) (organic vs. conventional) are shown. No consideration has been taken to the time since transition, landscape context or study region. Stars indicate significant differences between farming systems ($p\leq 0.05$).

		Farming system		Difference (%)
		<u>Organic</u>	<u>Conventional</u>	
Paper I	Butterfly richness	10.3 \pm 0.4	8.4 \pm 0.3	22.4*
($n_{\text{conv}}=20$)	Butterfly abundance	68.0 \pm 5.5	42.4 \pm 2.7	62.4*
($n_{\text{org}}=40$)	Plant richness	42.3 \pm 0.7	36.1 \pm 1.1	17.1*
Paper III	Moth richness	19.6 \pm 2.4	14.7 \pm 3.1	33.5
($n_{\text{conv}}=6$)	Moth abundance	22.8 \pm 56.2	14.8 \pm 80.2	54.4
($n_{\text{org}}=12$)	Plant richness	17.8 \pm 1.1	17.3 \pm 2.1	2.4
	Plant frequency	111.5 \pm 6.5	94.0 \pm 7.4	22.9
Paper IV	Carabid richness	3.4 \pm 0.2	3.0 \pm 0.3	12.7
($n_{\text{conv}}=12$)	Carabid activity density	11.8 \pm 1.7	8.9 \pm 2.4	32.0
($n_{\text{org}}=24$)	Seed predation (%)	65.8 \pm 3.1	66.5 \pm 5.4	-1.1

Neither the carabids nor the rate of seed predation benefited from organic farming (paper IV). Although carabids may be favored by organic management, e.g. organic fertilization (Kromp, 1999), the composition of the surrounding landscape seems to influence the carabids more than the local farming practices (Östman *et al.*, 2001; Purtauf *et al.*, 2005; Winqvist *et al.*, 2011). Carabids are also tolerant against disturbances, making them less affected by a specific farming system (Thiele, 1977). This may explain the high variability among studies where benefits on carabids, if any, has been shown for both organic (Fischer *et al.*, 2011) and conventional farming (Weibull *et al.*, 2003; Ekroos *et al.*, 2010).

The relative difference in moth, plant and carabid diversity between organic and conventional farms in paper III and IV corresponds to the relative difference found for plants and butterflies in paper I. However, the difference was only statistically verified in paper I (Table 1). The fewer data points (i.e. farms) in paper III and IV, resulting in lower statistical power, can have decreased the chances of detecting a significant difference (i.e. type II error).

In addition to the time since transition and landscape context, the effect of organic farming may also depend on the conventional farmers' decision on using, or not using, agrochemicals. For example, organic farmers are not allowed to apply synthetic fertilizers or pesticides on their fields, but this does not necessarily entail that conventional farmers do. They decide themselves upon using these agrochemicals or not, and to what extent, and have thereby

the ability to equalize the differences between the farming systems. Also, some conventional farmers may have adopted agri-environment schemes other than organic farming that increase the quality of the surrounding non-crop habitats. As both landscape context and the time since transition have been controlled for in the study designs, the individuality of the farmers may have contributed to the high variability found in the effect of organic farming.

4.1.2 Species traits

The effect of organic farming on species traits was only tested for in paper II where no distinction among butterflies differing in dispersal capacity, reproductive rate or host plant specificity could be found. These results indicate that organic farming benefits all butterflies irrespective of traits, as the overall effect of organic farming was positive (paper I). Paper III showed that the effect of organic farming sometimes varies among species within the same species group. This is likely the result of species specific traits, or presumably a combination of traits, that makes certain species more susceptible to organic farming than others. This is, however, difficult to test given the ambiguity of species traits and that traits typically are correlated (but see Öckinger *et al.*, 2010).

4.2 Effects of the time since transition to organic farming

Not all species displayed temporal effects to organic farming, but some did. Either way, the information provided is as equally important due to its novelty, and help provide deeper insights into species responses and the effect of organic farming over time.

In paper I, plant and butterfly species richness increased rapidly at the transition to organic farming, but did not continue to increase the following years. In contrast, butterfly abundance experienced a gradual increase and farms that had been organic for 25 years had on average 100% higher abundance of butterflies compared to newly transitioned farms (Figure 5). These responses could not be explained by species traits (paper II).

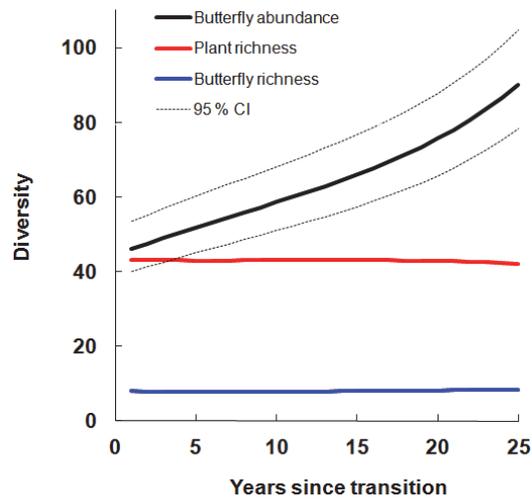


Figure 5. Plant species richness and butterfly species richness and abundance in relation to the time since transition to organic farming. The figure is based on model average parameter estimates.

The overall effect of organic farming on moth species richness and abundance was not significant. However, when the farms were categorized as “new” and “old” organic the analyses unraveled a different result; the moths increased in richness and abundance only on the new organic farms ($TST \leq 6$ years), whereas conventional and old organic farms ($TST \geq 15$ years) had similar diversity (paper III; Figure 6a and b). This result clearly shows that depending on the time since transition, the effect of organic farming can be both positive or neutral, or possibly even negative. The plant community displayed a similar, but non-significant, pattern with higher plant richness and frequency on the new organic farms (Figure 6c and d). However, four species stood out when the analyses were made at the species level; *Cirsium arvense* and *Stellaria media* had significantly higher frequency on new organic farms and *Equisetum pratense* and *Poa pratensis* on old organic farms.

Neither the carabids nor the ecosystem service of weed seed predation displayed any temporal effects.

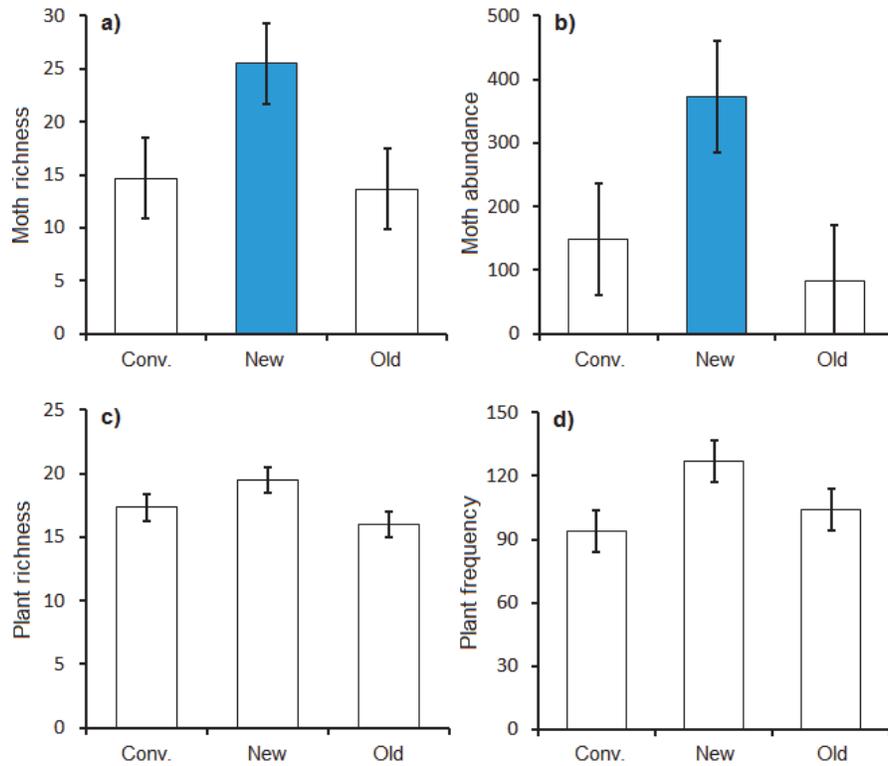


Figure 6. Moth species richness (a) and abundance (b) and plant species richness (c) and frequency (d) on conventional, new, and old organic farms. Different colors indicate statistically significant difference at $p \leq 0.05$.

4.2.1 Why temporal effects?

The underlying mechanisms to the temporal effects need additional attention, but the dynamics of plant diversity seemed to have high explanatory effect for butterflies and moths. Arable land generally has low carrying capacity (i.e. the maximum population size the environment can hold, given the availability of essential resources) for plant dependent species such as butterflies and moths. In paper I, the plants had higher species richness on organic farms but they did not increase with the time since transition as the butterfly abundance did. The increased plant species richness could, however, have constituted the extra resources needed for the butterflies to be able to step by step increase in population size. An alternative explanation could be that the size of the species pool is similar today as it was 25 years ago, but that the population sizes on conventional farms by time have decreased as a result of an overall lower habitat and resource availability at landscape scales. If so, it does not necessary mean that organic farming can boost the populations, as interpreted by the

significant temporal effect, but rather that organic farms have conserved the population sizes seen during the particular year of transition. Unfortunately, no such long term time series are available for the organic farms, but landscape analyses using maps derived from the year of transition could perhaps help disentangle this question.

The moths' responses seem to be driven mainly by one single plant species, *Cirsium arvense* (Figure 7). *C. arvense* is a problematic weed for organic farmers, but is highly preferred by several nectar and pollen feeding insects (Alanen *et al.*, 2011). Individuals of *C. arvense* can develop quickly after a disturbance thanks to a deep and highly regenerative root system (Gustavsson, 1997). Hence, before the indirect weed control methods used on organic fields (e.g. varied crop rotations) start to be effective, *C. arvense* has the ability to develop large populations in the absence of herbicides. This explains the higher frequency of *C. arvense* found on the newly transitioned organic farms, and to the subsequent response of the moths. It is unfortunate that the moth data only derive from one point in time, corresponding to the flowering period of *C. arvense*. It can therefore not be verified if the moths only experienced temporal effects to organic farming as a result of the presence of *C. arvense*, or if temporal variation to organic farming existed at times when *C. arvense* was not in bloom. This question needs to be addressed further.



Figure 7. The Large Skipper (*Ochlodes sylvanus*) on a Creeping Thistle (*Cirsium arvense*). (Photo: Dennis Jonason)

4.3 Effects of landscape composition

4.3.1 Biodiversity

Landscapes with increasing proportion of arable land had lower species richness of butterflies, but it was still higher on organic compared to conventional farms (paper I; see also section 4.3.2).

No landscape effect was found on moths or plants (paper III). Moths are strong flyers (Betzholtz & Franzén, 2011) which make them less susceptible to large scale landscape simplification, whereas plants in general are considered sedentary. Sedentary species are more affected by the nearby surroundings, and although landscape compositions at different scales are more or less correlated, the 1 km scale used in the studies can consequently have been too large for landscape composition to affect the plants. Moreover, plants respond slowly to landscape changes (e.g. Lindborg & Eriksson, 2004), making it complicated to distinguish any relationship to present day landscape structure. Complementary landscape analyses using historical maps could possibly reveal other information and may come up for consideration in future studies.

Many carabid species originate from the past-glacial primarily open and dry habitats and feed, reproduce and overwinter in arable fields (Thiele, 1977; Andersen & Eltun, 2000). This contrasts with most species in agricultural landscapes, but likely explains the fact that carabid species richness was positively related to the area of annual crops (paper IV) (the area of annual crops is highly correlated to the proportion of arable land which was used as proxy for landscape heterogeneity in paper I-III). This was, however, only evident in Uppland and not in Scania, nor was it evident for carabid activity density. The evenness of species, i.e. the relative abundances of species in a community, decreased with decreasing landscape heterogeneity, implying that some species were favored more than others by large total area of annual crops. Based on these results and existing literature, it can be concluded that the distribution of carabids in agricultural landscapes is hard to predict. Landscape composition seems to have an effect, but whereas this study found a positive relationship to the area of annual crops, other studies have found the opposite result (Weibull *et al.*, 2003; Purtauf *et al.*, 2005).

4.3.2 Species traits

In contrast to the observation that organic farms had higher butterfly species richness compared to conventional farms irrespective of landscape context, the species pool in the homogeneous landscapes was dominated by species with high mobility and reproduction, causing homogenization of the butterfly community (paper II). High mobility species have the ability to trace scattered

resources and high reproductive species can buffer for periods of low resource availability while taking advantage of the resources once they become available, and thereby give the population a sudden boost. When communities become homogenized, the functioning of ecosystems may be reduced as trait diversity largely determines the range of functions performed (Diaz & Cabido, 2001). Hence, although organic farming increased the butterfly species richness, the trait diversity did not increase and it was impaired by landscape simplification. This indicates that organic farming applied at the local scale cannot counteract landscape effects at larger scales. To enhance the wider ecological and environmental benefits of organic farming, policy changes need to be made towards a landscape scale targeting.

4.3.3 Weed seed predation

The number of predated seeds was positively related to carabid species richness and to the area of annual crops. Surprisingly, Uppland had lower predation rates compared to Scania despite 250% higher activity density. Other granivorous (seed eating) species may possibly have contributed to the seed predation in homogeneous landscapes, but this is rather unlikely given the design of the seed cups and the lack of slugs and ants in the pit fall catches. The regional differences may instead be due to different community compositions. In Scania, but not in Uppland, the genus *Pterostichus* had largest positive relationship to seed predation despite high abundances in both regions ($R = 0.51$, $p = 0.031$). In contrast, the seed predation in Uppland was related to *Trechus* spp. ($R = 0.44$, $p = 0.070$). *Trechus quadristriatus* from this genus has been shown to have a substantially higher seed consumption rate compared to many other species (Honek *et al.*, 2003). Hence, the high activity density of *T. quadristriatus* in Uppland but not in Scania (191 and 27 individuals, respectively) may explain the different relationship to seed predation between regions.

The average seed predation across the study was 66%. Seed mortality below that level has in modelling studies been shown to cause negative effects on weed populations (e.g. Westerman *et al.*, 2005), supporting the notion that weed seed predation can be used as biological control to reduce to usage of herbicides. Unfortunately, several of the most notorious weeds propagate not only from seeds, but also from roots and rhizomes (e.g. *Elytrigia repens*, *Cirsium arvense* and *Sonchus arvensis*). The actual contribution of seed predators to reduce the weed problem may therefore be limited, but needs further attention.

A consideration of species traits, as opposed to species richness or activity density, would be valuable for better understanding of carabids' responses to

agricultural practices, landscape context and to their delivery of ecosystem services (Winqvist *et al.*, 2011; see also paper II). For instance, differences in diurnal activity patterns (Thiele, 1977) make day-active species more at risk for pesticide spraying compared to night-active species and carabid body mass is related to the mass of preferred seeds (Honek *et al.*, 2007, but see Brose *et al.*, 2008).

4.4 Interaction effects between the time since transition and landscape composition

The hypothesis that biodiversity would respond faster to organic farming in heterogeneous landscapes, given a larger species pool and shorter distances from source habitats, could not be supported in any of the papers. Landscape heterogeneity was measured as the proportion of arable land within a 1 km radius and was foremost used to aid the selection of farms. This landscape measure gives a fair estimation of the degree of potential source habitats within 1 km, but it does not tell about where in space these are located. The hypothesis of an interaction effect between the time since transition and landscape composition wrongfully assumes that all source habitats in homogeneous landscape are located far from the place of data collection, whereas they hypothetically can be more proximate there than in heterogeneous landscapes. Hence, species responses to organic farming can be fast even in the most homogeneous landscapes if the proximity to source habitats is short. A more appropriate and biologically meaningful method to test the hypothesis would have been to map the distances from potential source habitats to the organic field and integrate that into a landscape resistance measure (*sensu* Wasserman *et al.*, 2012), alternatively to collect data in landscapes differing in heterogeneity but at a fixed distance from source habitats.

The lack of interaction effect may also be due to that the studied landscapes were not homogeneous enough or due to that the 1 km scale was too narrow, as most species evidently have been able to colonize the organic fields irrespective of landscape composition. The hypothesis is probably more likely to be confirmed if the matrix surrounding each sampling location is truly hostile and not semi-hostile. Although (conventional) arable fields may be considered as hostile, they are not hostile in the same sense as for example water, at least not in this part of the world.

5 Conclusions and implications

This thesis has shown that organic farming has a temporal component that can reveal unforeseen information in evaluations of the effect on biodiversity. It will be essential that this component is acknowledged in future evaluations of organic farming, otherwise it can obscure the interpretation of the results and obstruct the development of new policies aiming for sustainable agriculture and farmland conservation.

5.1 Future challenges of organic farming

Agri-environment schemes (AES) have been suggested to be the only feasible way to counteract the declining biodiversity in agricultural landscapes (Donald & Evans, 2006), yet their efficiency has been questioned (Kleijn & Sutherland, 2003; Kleijn *et al.*, 2006). Whereas agricultural intensification acts at both small and large scales (Benton *et al.*, 2003), the uptake of organic farming is mainly made at the field or farm scale (Whittingham, 2007). It is therefore likely that some of the biodiversity benefits will not be realized simply because species respond to and utilize the landscape at larger scales than the scale of an average sized farm. To improve the efficiency of AESs such as organic farming, there is a prevalent consensus in favor of policy changes towards a targeted landscape-scale approach (Gabriel *et al.*, 2009, 2010; Merckx *et al.*, 2009; Concepción *et al.*, 2012; paper II). How this should be achieved and to what extent it promotes biodiversity needs to be investigated further.

Geiger *et al.* (2010) disentangled the negative effects of agricultural intensification on biodiversity, i.e. arguing that it was mainly caused by pesticides, but it remains unclear what is causing the positive effects of organic farming. Compared to other species that have shown both positive and neutral responses to organic farming, plant diversity benefits repeatedly (Bengtsson *et al.*, 2005). Plants constitute a fundamental part of the agroecosystem by giving rise to a variety of resources, and an increase or a reduction in this resource

may result in knock-on effects at both higher and lower trophic levels (Hawes *et al.*, 2003). If it is the high weed levels in organic farming that constitute the major benefits for biodiversity, as proposed in paper III and in paper II in part, the question is how the constant strive towards more efficient non-chemical weed control technologies (Van Der Weide *et al.*, 2008) will affect the benefits of organic farming in the long run. To combine production with conservation will be a serious challenge for the future.

The principles of organic agriculture state that “production is to be based on ecological processes” (IFOAM, 2012). Despite this, the words biodiversity, species richness or ecosystem services are not mentioned in Commission Regulation No 889/2008 (EC, 2008) that lays down detailed rules on organic production. To be able to increase the yields in absence of synthetic fertilizers and pesticides, the focus of organic farming cannot solely be put on the arable field. More efforts need to be placed on restoration, creation and management of non-crop habitats such that these can help sustain the populations of beneficial species groups, either by cooperating with other AES that do so or by introducing new regulations and policy measures in favor of this.

Organic farming has the potential to counteract the negative effects caused by the agricultural intensification on biodiversity and environment, but opportunities for improvements evidently still exist. Acknowledging temporal effects is one step in the right direction.

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Svensk sammanfattning (Swedish summary)

Efterkrigstidens kraftiga befolkningstillväxt och teknikutveckling orsakade stora förändringar inom jordbruket, där det traditionella och småskaliga brukandet av jorden fick ge vika för mer moderna och effektiva metoder. Denna jordbrukets intensifiering har kraftigt försämrat levnadsvillkoren för många djur och växter i jordbrukslandskapet och är en av de enskilt största bidragande orsakerna till det hot vi idag ser mot den biologiska mångfalden. Intensifieringen består av flera komponenter, varav omvandlingen av naturliga livsmiljöer till åkermark och användandet av konstgödsel och syntetiska bekämpningsmedel tillhör de mest utmärkande.

Flertalet miljöersättningar ersätter idag lantbrukare för kostnader relaterade till miljöförbättrande åtgärder, med syfte att motverka jordbrukets intensifiering. En av miljöersättningarna är ekologisk odling, där användningen av konstgödsel och syntetiska bekämpningsmedel är förbjuden. Detta anses vara en central förklaring till de positiva effekter på den biologiska mångfalden som uppvisats i förhållande till konventionellt jordbruk.

Hur effekten på den biologiska mångfalden speglar sig i utvärderingar påverkas av ett flertal faktorer, bortsett från den ekologiska odlingen i sig. Landskapets utformning är exempelvis av stor betydelse. Jordbrukslandskap med ett varierat inslag av åkrar, betsmarker, skogar, våtmarker, etc., har i regel en relativt hög artdiversitet tack vare tillgängligheten av olika livsmiljöer. Detta till skillnad mot jordbrukslandskap enbart bestående av åkermark. Ekologisk odling har således ansetts få låg relativ effekt i heterogena landskap där artdiversiteten redan är hög och inte kan öka något nämnvärt, liksom i homogena landskap på grund av att artpoolen är så liten och avstånden till möjliga spridningskällor är så stora. Alltså, beroende på i vilka typer av landskap gårdarna ligger så kan effekten av ekologisk odling skilja sig åt, med generellt högst effekt i landskap av genomsnittlig komplexitet.

Tiden sedan omställning från konventionell odling har även den ansetts kunna vara av betydelse för effekten av ekologisk odling på den biologiska mångfalden. Man känner till från andra typer av studier att arters respons inte alltid sker parallellt med miljöförändringarna de utsätts för, men trots det har tidsfaktorn nästan helt och hållet bortsetts från i utvärderingar av ekologisk odling. Det är den enskilt största orsaken till varför denna avhandling har gjorts. I och med att omställningen till ekologisk odling medför ett stopp av användandet av konstgödsel och syntetiska bekämpningsmedel anses förutsättningarna för en högre artrikedom öka. Detta betyder som sagt inte att ökningen av antalet arter nödvändigtvis sker omedelbart efter omställningen, utan snarare i takt med att följderna av de nya bruksmetoderna får fäste samt i takt med att nya arter hittar dit. Det senare kan i sin tur även det påverkas av omgivande landskap, då närheten till andra habitat, samt arters spridningsförmåga, kan avgöra vilken tidsram det handlar om.

Jag har i min avhandling testat effekten av ekologisk odling på dagfjärilar, nattfjärilar, växter, jordlöpare, samt på ekosystemtjänsten ogräsfröpredation. Data har samlats in på konventionella och ekologiska gårdar lokaliserade i landskap av olika komplexitet. De ekologiska gårdarna har haft olika tid sedan omställning (1-25 år). Utifrån detta upplägg har jag haft möjligheten att analysera effekten av ekologisk odling och tiden sedan omställning med hänsyn till landskapets komposition.

Det fanns ingen entydig effekt av ekologisk odling. Antalet dagfjärilsarter ökade med 20 %. Det gjorde även antalet växtarter, men dock bara i en av två studier. För växterna i den andra studien, samt för nattfjärilarna, jordlöparna och fröpredationen, var effekten snarast neutral. Tittade man däremot närmare på hur effekten av ekologisk odling såg ut över tid hittades intressanta och uppseendeväckande resultat. Antalet dagfjärilsindivider ökade gradvis med tiden sedan omställning för att efter 25 år vara 100 % högre jämfört med år ett, medan artrikedomen av dagfjärilar och växter ökade i direkt anslutning till omställningen för att sedan ligga på en jämn nivå. Nattfjärilarna som tycktes vara oberoende av odlingssystem visade en tydlig positiv reaktion på nyomställda gårdar (tid sedan omställning ≤ 6 år) medan konventionella och äldre ekologiska gårdar (≥ 15 år) hade lika art- och individrikedom. Även för växterna som inte reagerade på ekologisk odling fanns en tidseffekt, men bara på artnivå och inte för hela växtsamhället. Åkertistel (*Cirsium arvense*) och våtarv (*Stellaria media*) var mer frekventa på nya ekologiska gårdar medan ängsfräken (*Equisetum pratense*) och ängsgröe (*Poa pratensis*) var mer frekventa på äldre ekologisk gårdar. Den höga frekvensen av framför allt åkertistel tros orsakas av att de ekologiska metoderna för att hantera ogräs utan bekämpningsmedel (ex. hög grödrotation med stort inslag av vall) kräver tid

för att få maximal genomslagskraft. Åkertistel, som från små rotfragment lätt kan producera nya individer, utnyttjar detta och kan således snabbt öka i antal på nyomställda gårdar där växtskyddet är som lägst. För lantbrukarna utgör åkertistel ett stort problem, medan den är mycket viktig som nektar- och pollenkälla för många insekter i jordbrukslandskapet. Jag tror därför att åkertistel kan vara en stor bidragande orsak till nattfjärilarnas respons som följde samma mönster. Effekten av ekologisk odling och tiden sedan omställning var oberoende av landskapets komposition. Ingen tidseffekt noterades för varken jordlöparna eller för ogräsfröpredationen.

Utvärderingar lägger grunden för hur miljöersättningar och deras utformning ska kunna effektiviseras, inte bara vad gäller förbättrat utfall på miljö och biologisk mångfald utan även vad gäller minskade kostnader. Det är därför av yttersta vikt att utvärderingar förmedlar en så övergripande bild som möjligt. Med hjälp av den här avhandlingen har jag visat att det finns en viktig och tidigare ostuderad temporal effekt av ekologisk odling. Hänsyn till denna faktor i framtida utvärderingar kommer att leda till högre tillförlitlighet och en ökad förståelse av samspelet mellan ekologisk odling, biologisk mångfald och ekosystemtjänster över tid. Framtida forskning bör fokusera på att försöka identifiera de underliggande mekanismer för hur ekologisk odling och andra miljöersättningar påverkar olika arter och funktioner, för att på så sätt kunna rikta insatser för en snabbare och mer långvarig effekt.

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