

CHAPTER 6

Does organic farming affect biodiversity?

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Let us give Nature a chance; she knows her business better than we do.

Michel de Montaigne

6.1 Introduction

The second half of the twentieth century has seen enormous losses of farmland biodiversity in the UK. The consensus is that agricultural intensification is largely responsible (Chapter 1, this volume). For example, European countries with higher wheat yields tend to have more threatened or recently extinct arable weed species (Storkey et al. 2012). Similarly, plant species richness in six European countries has been shown to be negatively related to nitrogen input (Kleijn et al. 2009). For birds, population declines and range contractions are greatest in European countries which have the highest levels of production, with cereal yields indicating a suite of crop and livestock husbandry changes (Donald et al. 2001, 2006). These trends appear set to continue, given ever greater pressures on the land, exerted by the expansion of human populations and the need for food security, compounded by issues such as globalization of markets, biofuels, and climate change (Tilman et al. 2001). Non-cropped habitats on farmland, such as hedgerows, ponds, and patches of woodland, have also fared badly, with periods when they were aggressively removed to increase the cultivated area (Chapter 1, this volume). For example, in 1946, there were around 800 000 km of hedgerow in Britain, of which 1300–5500 km were destroyed annually between 1946 and 1970 (O'Connor et al. 1986). Since 1930, 7% of the UK's ancient woodland has been lost completely, while 38% has been replaced with plantations of non-native species (Spencer and Kirby 1992).

Intensification brings increases in the use of pesticides and fertilizers, changes in cropping patterns (such as a shift from spring sown to winter sown crops, and a reduction in the use of traditional crop rotations), land drainage, pasture improvement, and reduction or loss of quality of non-cropped habitats such as hedgerows (Fuller 1987; Vickery et al. 2001). While there is broad agreement that agricultural intensification has largely been responsible for the widespread declines of much farmland wildlife (the evidence for farmland birds is particularly strong: Chamberlain et al. 2000; Donald et al. 2001; Wilson et al. 2009), disentangling the underlying causes is challenging. Some studies have attempted this: for example, Geiger et al. (2010) measured 13 components of intensification and found that the use of insecticides and fungicides has had the most consistent negative effects on biodiversity. Chamberlain et al. (2000) speculated that there may have been a threshold for critical amounts of food and habitat, and that this threshold could explain the lag they observed between changes in land use and the effect on bird populations.

Agricultural production and biodiversity conservation place differing demands on the land, and have provoked much debate about farming 'systems'. The term 'system' has come to represent different philosophies about the long-term sustainability of food production and biodiversity conservation. The issue is not straightforwardly one of direct conflict between biodiversity and crop production—it is more subtle than

that—but if an increasing percentage of primary production is channelled into crop production, then, all else being equal, biodiversity will suffer. Differences between ‘land sharing’ and ‘land sparing’ approaches have dominated much of the debate. At one end of the spectrum, land sparing advocates intensive management to maximize production in some areas while sparing others for the conservation of biodiversity. At the other end, land sharing implies the use of less intensive farming practice throughout much of the landscape, its philosophy being to encourage biodiversity as an integral part of, economically viable, farming (Green et al. 2005). Broadly speaking, agri-environment schemes (which offer financial incentives to farmers in the UK and Europe for restoring or managing habitat features for wildlife on farmland: Chapter 1, this volume) fall within this end of the spectrum, although there is much debate concerning their effectiveness e.g. Kleijn et al. (2006). Recently, the concept of sustainable intensification has gained momentum. Defined as a form of production wherein ‘yields are increased without adverse environmental impact and without the cultivation of more land’, it denotes an aspiration of what needs to be achieved (Garnet and Godfray 2012), and could be considered a useful framework for deciding which combinations of approaches might work best.

Organic farming is one form of low intensity farming which has attracted considerable attention. It relies on management using crop rotations (usually incorporating fertility-building grass leys), green manure, compost, and biological pest control. There is a greater emphasis on spring sowing of cereals (with associated overwinter stubbles). Fertilizers and pesticides are used, but they are strictly limited in type and amount compared with more non-organic farming. Non-organic farming mainly uses mineral fertilizers. These are not permitted on organic farms and animal manures are therefore important, hence many organic enterprises are mixed farms, having both arable and livestock. Certifying bodies hold lists of other products which may be applied, and prior permission may be necessary before some of these can be used. Organic farming is an internationally recognized system, with national standards for what can be labelled as ‘organic’ produce, based on the standards set by the International Federation of Organic Agriculture Movements (IFOAM), an umbrella organization established in 1972. In the UK, the Soil Association (SA) sets the standards that organic farmers have to adhere to in order for organic status to be conferred. The certification is important—the increased cost of organic production has been largely offset for the farmer by the premium price

that consumers have hitherto been willing to pay. The SA emphasizes that the organic system is ‘holistic’—its standards taking animal welfare and the environment, as well as farm management *per se*, into account.

Organic farming and the ‘extensification’ that is part of its philosophy often comes at a cost to production. For example, Gabriel et al. (2010) found organic crop yields to be around 55% of those of non-organic farms in two areas of England. Yield differences between organic and non-organic farms depend on site characteristics (Seufert et al. 2012), but the generally lower yields from organic farming have led to doubts that it can make a meaningful contribution to resolving the wildlife/production conflict over large areas (Gabriel et al. 2010). How to balance areas of wildlife-friendly land versus productive land under different crop yield scenarios has been explored by Hodgson et al. (2010), who predicted that farming non-organically, while sparing land as nature reserves, would be better for butterflies than farming organically over the same area, as soon as the organic yield per hectare fell below approximately 90% of the non-organic yield.

The scale at which trade-offs should be measured is not straightforward, though. For example, what if increased local biodiversity is at the cost of displacing the impacts on biodiversity possibly to somewhere else where production is increased, with even more severe consequences? Furthermore, the landscape context is important. Some studies have demonstrated that the benefits to biodiversity from less intensive (usually organic) farms are greater in simple, less heterogeneous landscapes (e.g. Roschewitz et al. 2005; Rundlof and Smith 2006), while other work points to the benefits of concentrating support for lower intensity farms in areas that already retain higher levels of biodiversity (Kleijn et al. 2009); more analyses are needed across broad spatial scales and over the long term (Balmford et al. 2012).

An obvious approach for tackling questions about the effects of intensification is to make comparisons between farming systems which differ in their intensity, while controlling for landscape context. Organic farms therefore provide an opportunity for carrying out an observational experiment on farming intensification in the UK. If organic farms have fewer aspects typical of more intensive systems, is there any evidence that they benefit biodiversity? A further issue surrounding organic farming and biodiversity is whether forms of non-organic extensification can deliver any biodiversity benefits associated with organic management. There is a continuum of possibilities

here that range from extensification of the entire farm area through, for example, low intensity grazing and various low input systems, to fine-scale land sparing. In the latter, intensive production over, say, 95% of the land area and a 5% sacrificial area where a farmer might create semi-natural habitat on less productive land would not necessarily support 'farmland' wildlife, but might be better for biodiversity than a land sharing approach, of which organic farming is arguably an example.

In the early 1990s, interest in the potential of organic farming for biodiversity conservation was growing. In 1994, WildCRU were invited to participate in a unique collaboration between the Research Councils. Biology, sociology, and economics were brought together to explore all aspects of organic farming systems, with the work based at HRH Prince Charles's organic farm in Gloucestershire (Cobb et al. 1999). WildCRU, together with colleagues from the Centre for Ecology and Hydrology (CEH), were responsible for quantifying biodiversity differences between organic and non-organic farms.

We decided to focus on two groups of invertebrates: butterflies and spiders. Butterflies have complex life cycles, comprising egg, larval, pupal and adult stages, and thus requiring larval foodplants (often species specific), a continuity of nectar sources for the adult stages during the spring and summer, and safe overwintering habitats. Their demands for specific microclimate, vegetation structure, and the co-occurrence of specific vegetation types suggest they can be useful indicators of some aspects of habitat quality on farmland (Erhardt and Thomas 1991). Butterflies respond rapidly to changes in plant communities (Feber et al. 1996) and, as nectar-feeders, are also likely to respond to the effects of farm management in similar ways to the wider group of wild pollinators, many of which are in severe decline (Potts et al. 2010). Butterflies are well-monitored in Britain and elsewhere, and could be considered as indicators of change in wider terrestrial invertebrates (Thomas 2005). The evidence for steep population declines in British butterflies is unequivocal: Fox et al. (2011) reported that 72% of butterfly species have decreased in abundance over 10 years and 54% have decreased in distribution at the UK level and, notably, the abundance of common butterflies had dropped by 24% over ten years. Loss of habitat due to urbanization, lack of woodland management, and intensification of farmland are all likely to have contributed to these declines. On farmland, key factors are likely to be loss of larval foodplants and adult nectar sources

through grassland improvement (Vickery et al. 2001), herbicide use and fertilizer application (Smart et al. 2000), and reduction in uncropped habitats (Feber and Smith 1995).

Our other chosen group, spiders, are beneficial predatory invertebrates and highly important bio-control agents on farmland (Nyffeler and Sunderland 2003). Evidence from other studies suggested that they were likely to respond to different farmland features, such as architectural complexity of the vegetation (e.g. Baines et al. 1998; Chapter 3, this volume). Some aspects of organic management could plausibly increase the abundance of their prey—the use of manure for fertility, for example, and few or no inputs of pesticides (e.g. Haughton et al. 1999). Management of cropped, as well as uncropped, areas is likely to be important for spiders, many species of which can colonize crops rapidly and in great abundance when prey is plentiful. In arable areas, spiders will overwinter in uncropped habitats such as field margins and disperse from these areas into fields in the spring and summer (Lemke and Poehling 2002; Schmidt and Tscharrntke 2005; Öberg and Ekbohm 2006). Spiders are also affected by wider impacts such as landscape heterogeneity (Schmidt et al. 2005).

6.2 Impacts of organic farming on butterflies

To test whether butterflies differed in their abundance and species richness between the two farming systems, we studied 12 pairs of organic and non-organic farms across southern England. These farms were matched so as to eliminate geographic variation—the non-organic farms were neighbours of the organic farms and growing similar crops. Butterflies were monitored over three years on these farms (Feber et al. 2007). Recorders walked a measured fixed transect route which was divided into sections corresponding to crop and/or boundary type. Each transect included an area of organic and an area of non-organic farmland (transect lengths within each farm pair were broadly similar). For each section, all butterflies seen were recorded; those seen associated with the crop edge were recorded separately from those associated with the uncropped field boundary. Details of crop, boundary type, and vegetation were recorded for each section of the transect routes (see Feber et al. 2007 for details).

Over the three years, we recorded 28 species of butterfly (Britain's butterfly fauna comprises around 56 breeding species). Averaged over each season as a

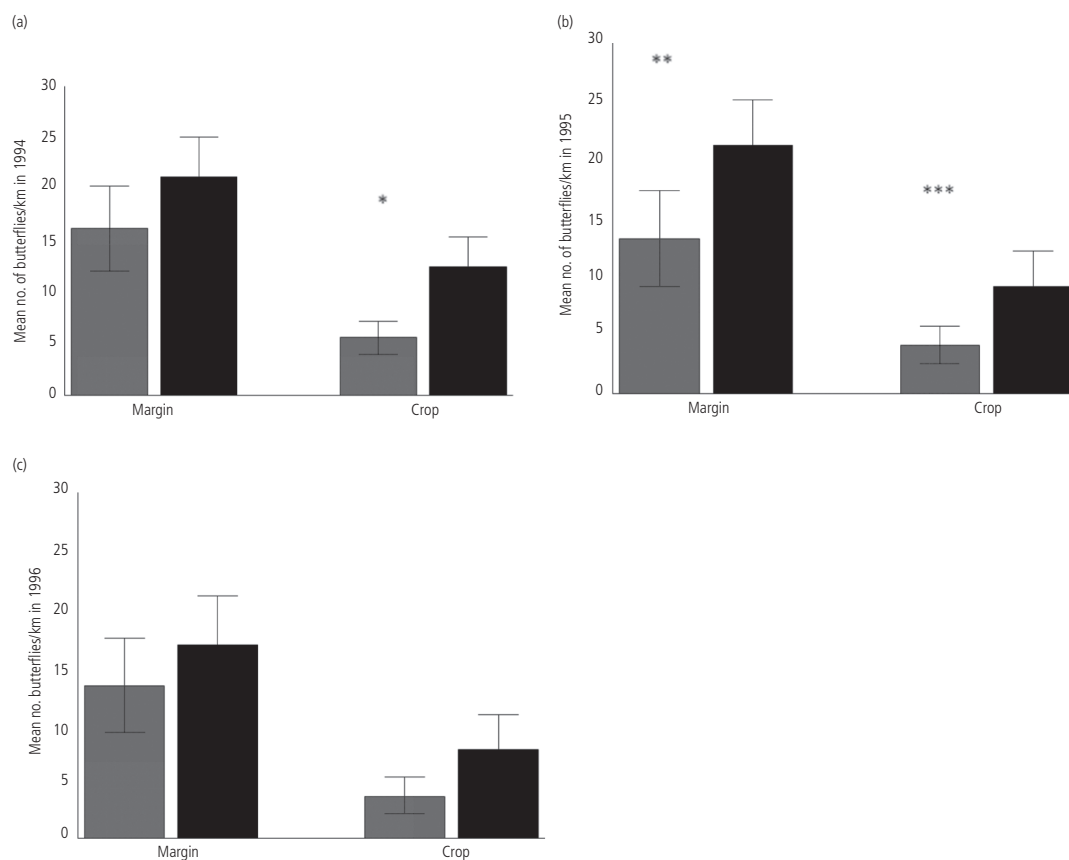


Figure 6.1 Effects of farming system (organic or non-organic) and habitat (margin or crop) on the mean abundance of butterflies per kilometre per transect in (a) 1994, (b) 1995, and (c) 1996. Solid bars: organic farmland, shaded bars: non-organic farmland. Significance of system effect indicated by asterisks. From Feber et al. (2007).

whole, we found that butterfly abundance was higher on organic compared to non-organic farms in all three years of the study (Fig. 6.1a,b,c, Table 6.1). Species richness followed a similar, but less striking, pattern, with more butterfly species on organic farms in each year, significantly so in 1994.¹ Most species had higher abundances of individuals recorded on organic than on non-organic farmland in most years, significantly so in at least one year for three of the less mobile species: large skipper *Ochlodes venata*,² common blue

Polyommatus icarus,³ and meadow brown *Maniola jurtina*.⁴ The large and small white butterflies (often known as ‘cabbage whites’, considered to be pest species of brassica crops) are typically more mobile and organic farming did not increase their abundance (Feber et al. 2007).

Butterflies were consistently more abundant on uncropped margins compared with the cropped area (Fig. 6.1), and there was also a tendency for the system effect to be more marked on the cropped area

¹ $t = 7.48$, d.f. = 7, $P < 0.001$: Feber et al. 2007.

² 1994: Mean abundance per km (S.E. in parentheses): 3.2 (0.98) on organic, 1.5 (0.66) on non-organic. $F_{(1,7)} = 6.17$, $P = 0.042$.

³ 1994: Mean abundance per km (S.E. in parentheses): 7.4 (2.33) on organic, 2.5 (0.98) on non-organic, $F_{(1,7)} = 6.95$, $P = 0.034$.

⁴ 1995: Mean abundance per km (S.E. in parentheses): 15.5 (1.54) on organic, 9.1 (1.07) on non-organic, $F_{(1,7)} = 21.51$, $P = 0.001$.

Table 6.1 ANOVA summary of the effects of farming system (organic or conventional) and habitat (margin or crop) on the mean abundance of butterflies (per kilometre, per transect) between 1994 and 1996. Analyses were performed on log-transformed data. Asterisks indicate level of significance of effect (exact *P* values also given). *n* = number of pairs of farms. From: Feber et al. (2007).

FACTOR	YEAR					
	1994 (<i>n</i> = 8)		1995 (<i>n</i> = 10)		1996 (<i>n</i> = 5)	
	<i>F</i> (1,7)	<i>P</i>	<i>F</i> (1,9)	<i>P</i>	<i>F</i> (1,4)	<i>P</i>
System	9.22	* 0.020	31.46	*** 0.001	9.51	* 0.037
Habitat	13.00	** 0.009	22.83	*** 0.001	8.28	* 0.045
System x Habitat (interaction term)	5.16	ns 0.057	6.40	* 0.032	0.68	ns 0.450

compared with the uncropped margin.⁵ The effect of system was therefore likely to be the result of differences in management of both uncropped and cropped habitats.

What might cause these differences in butterfly abundance and species richness? In intensively farmed landscapes, loss of plant diversity in hedge bottoms and grasslands has diminished the abundance and types of larval foodplants (many butterfly species, in their larval stage, are restricted to one or two species of plant) and reduced the availability of nectar sources for foraging adults (Feber and Smith 1995). Organic farms in our study had larger hedgerows and more perennial field-edge plant communities than non-organic farms, providing shelter and increased protection from pesticide application, as well as enhanced food resources; via these mechanisms they may have offered better habitat quality for farmland butterflies. Organic farms also differed from non-organic farms in terms of cropping regime. Because the use of artificial fertilizers is prohibited within organic systems, grass-clover leys usually form an integral part of the rotation to restore and maintain fertility. This increased proportion of grassland within organic farms is likely to have benefits for butterflies. Larvae of *M. jurtina* (meadow brown) and *Pyronia tithonus* (gatekeeper), for example, are grass-feeders, and these species require undisturbed swards in which to overwinter in their larval stage. It seems likely that the greater temporal stability resulting from the maintenance of grass leys, which are in place for two or more years, the presence of more perennial field edge plant communities, and an increase in spatial heterogeneity at a landscape level sustained a greater diversity and abundance of butterflies.

⁵ Indicated by a statistical interaction between the main effects of system (organic vs non-organic) and location (margin versus crop)—the effect was statistically significant in both 1994 and 1995).

6.3 Impacts of organic farming on spiders

On three of the farm pairs, we also tested whether effects of farming system could be detected on spider abundance and species richness, by sampling surface-active spiders from winter wheat fields at each of three of the sites, using pitfall trapping (Feber et al. 1998). Twelve pitfall traps were placed in a grid formation in each of the 18 fields under study, and were left out for a week in May and again in June. A number of vegetation measures were also recorded in each field. The methodology is described in full in Feber et al. (1998).

We identified 56 spider species from 8609 individuals in our pitfall trap samples, with most species belonging to the family Linyphiidae ('money' spiders, typical of agricultural land in the UK) (e.g. Fig. 6.2a). The Lycosidae (wolf spiders) were also well represented by *Pardosa* and *Trochosa* spp. (e.g. Fig. 6.2b).

Both the number of spiders captured and the species richness of spider samples were higher in organic than non-organic winter wheat fields (Fig. 6.3a,b). There was a pronounced difference between organic and non-organic fields in terms of understorey vegetation (both grasses and forbs), which was substantially more abundant on organic fields, and organic winter wheat was less dense than non-organic (fewer crop plants per square metre). Our most consistent finding was that there was an increased abundance and species richness of spiders in our samples, with increasing abundance of understorey vegetation within the crop, both overall and within each system, within each sampling session. For spiders, these system effects may have been mediated by the increased structural complexity within the crops on organic forms, increasing the opportunities for web-builders, and enhancing the availability of prey (Baines et al. 1998). The absence of agrochemicals and more complex crop rotations may also affect spider communities. Birkhofer et al. (2008)

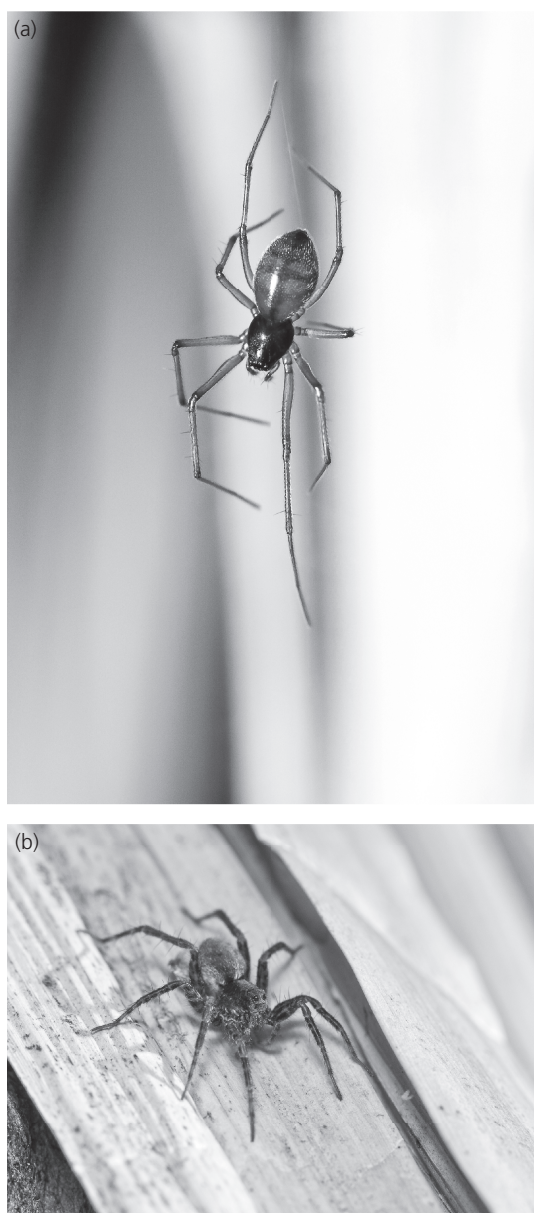


Figure 6.2 Spiders commonly occurring on arable farmland in our study included (a) members of the Linyphiidae ('money spiders') such as *Bathypantes gracilis* and (b) members of the Lycosidae ('wolf spiders') such as *Pardosa prativaga*. Photographs © Evan Jones.

found that organically managed grass-clover leys had higher spider activity density,⁶ and intensively farmed fields have been shown to have fewer spider species,

⁶ Activity density describes the number of individuals per unit captured by pitfall traps in a defined period (14 days for the study cited).

and lower activity of Lycosidae, than bio-dynamic fields (Gluck and Ingrish 1990).

6.4 Wider impacts of organic farming on biodiversity

A range of other studies have suggested that organic farming could benefit a wide range of taxa. There is evidence of positive effects of organic relative to non-organic for plants (e.g. Hyvonen et al. 2003; Gabriel et al 2006; Rundlof et al. 2010; Batáry et al. 2013), invertebrates (e.g. Holzschuh et al. 2008, Kragten et al. 2011), small mammals (e.g. Fischer et al. 2011), birds (e.g. Chamberlain et al. 1999, Kragten and de Snoo 2008, Smith et al. 2010), and bats (e.g. Wickramasinghe et al. 2003; Wickramasinghe et al. 2004). Meta-analysis of published studies using a range of methodologies and spatial scales (Bengtsson et al 2005) suggests that organic farming is associated with increased species richness and abundance of plants, predatory invertebrates, and birds and Hole et al.'s (2005) meta-analysis also provided clear evidence for a beneficial effect of organic farming on biodiversity. Tuck et al. (2014) found that, on average, organic farming increased species richness by about 30%. The response of animal communities to organic farming may partly depend on the trophic level of a taxonomic group (Birkhofer et al. 2014) and functional groups may be affected in different ways by organic farming (Batáry et al. 2012).

A shortcoming of many studies on the impacts of organic farming on biodiversity (reviewed by Hole et al., 2005), is that they have been limited in sample size or geographical scale. Furthermore, a common approach has been to pair out non-crop habitat differences between organic and non-organic farms and focus on system (within-field) effects alone, which prevents disentangling of the impacts of habitat features and field management on biodiversity. For example, the extent to which differences in biodiversity might be attributed to differences in the quality or quantity of uncropped habitat between organic and non-organic farms, or the extent to which field management might be the cause, remained poorly understood. This is important because, from a policy point of view, a key question remains—can non-organic farmers deploy elements of organic farms (such as amount or quality of uncropped habitats) which will result in biodiversity levels similar to those on organic farms? Or are there features intrinsic to the organic system (such as no artificial agrochemical input) that cannot easily be transferred to, or replicated within a non-organic system? As well as lack of pesticide and synthetic fertilizer use,

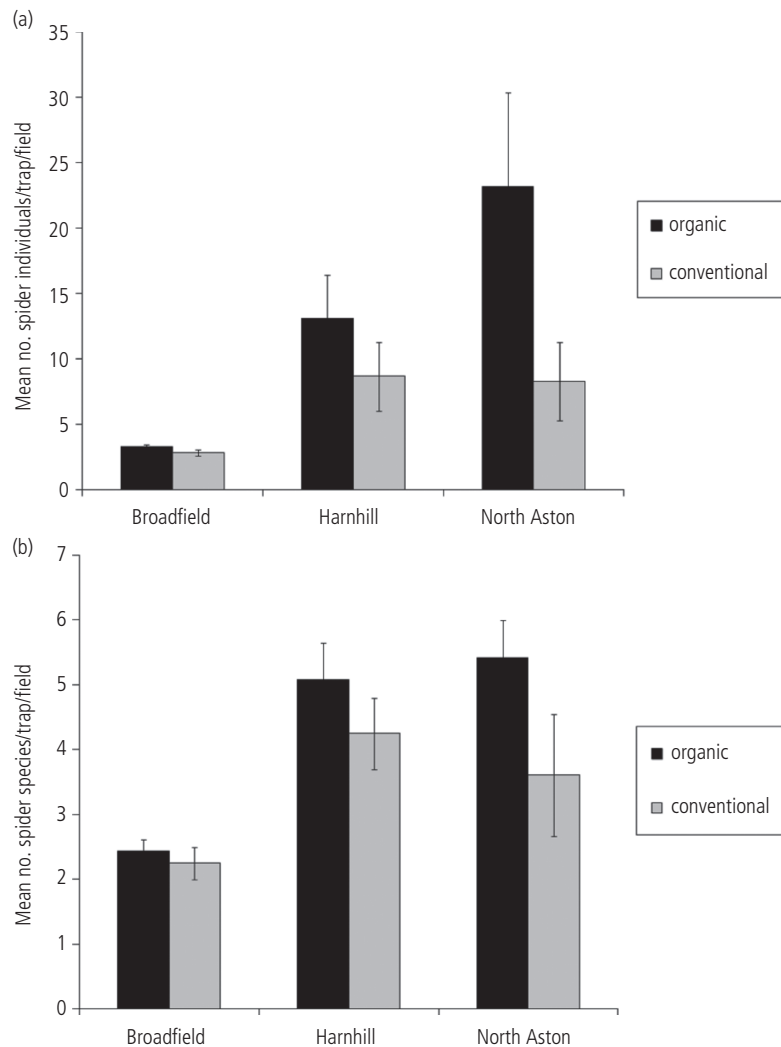


Figure 6.3 Mean number of (a) spider individuals and (b) spider species captured per trap per field on organic and non-organic (conventional) fields in June (see text for details). Re-drawn from Feber et al. (1998).

organic farms differ from non-organic farms in terms of a range of habitat variables and management practices (Norton et al. 2009), which vary in the extent to which they could be considered intrinsic to the system.

6.5 A large-scale multi-taxa study of organic farming

We used a multi-taxa, large-scale study of 89 pairs of farms (Fuller et al. 2005) to address these issues. Higher plants, spiders, ground (carabid) beetles, wintering birds, and bats were studied to represent a range

of trophic levels, niches, and ecological requirements. The project was of unprecedented scale for studies of this kind. Our project, a joint investigation with the British Trust for Ornithology (BTO), the Centre for Ecology and Hydrology (CEH), and the Royal Agricultural College (RAC), focused on the question: to what extent can biodiversity differences between organic and non-organic systems be related either to the amount and management of non-crop habitat, or to the differences in crop management?

The project had two primary aims. First, we aimed to assess the extent of differences in biodiversity between

organic and non-organic farming systems over a wide geographical scale for a large sample of farms. Second, we asked if the extent to which observed biodiversity differences could be attributed to features intrinsic to organic systems, such as lack of artificial agrochemical use, or whether there were elements that could be used in non-organic systems to benefit biodiversity (non-cropped habitats for example). Figure 6.4 illustrates some hypothetical links between system (S), habitat (H), and biodiversity (B). Both the biodiversity and habitat axes are purely notional, and the habitat variables in our thought experiment vary independently of the biodiversity axis in question. In reality it is unlikely that any habitat can vary without there being a

causal link with some aspect of biodiversity. In the first three scenarios (a, b, and c), no system effect on biodiversity is expected, as there is no direct or indirect link between S and B. In scenario (d), we would observe a habitat effect, but system would not have any additional effect; in this case, H would appear to be a predictor of B, but only in a statistical model which does not take account of S. In (e), an effect of system is detected but habitat would not be a useful predictor in any model under this scenario, regardless of the inclusion of system. In (f), statistical models would detect an additive effect of system and habitat. Our paired design would detect the system effect in models not including habitat. But the system effect is not

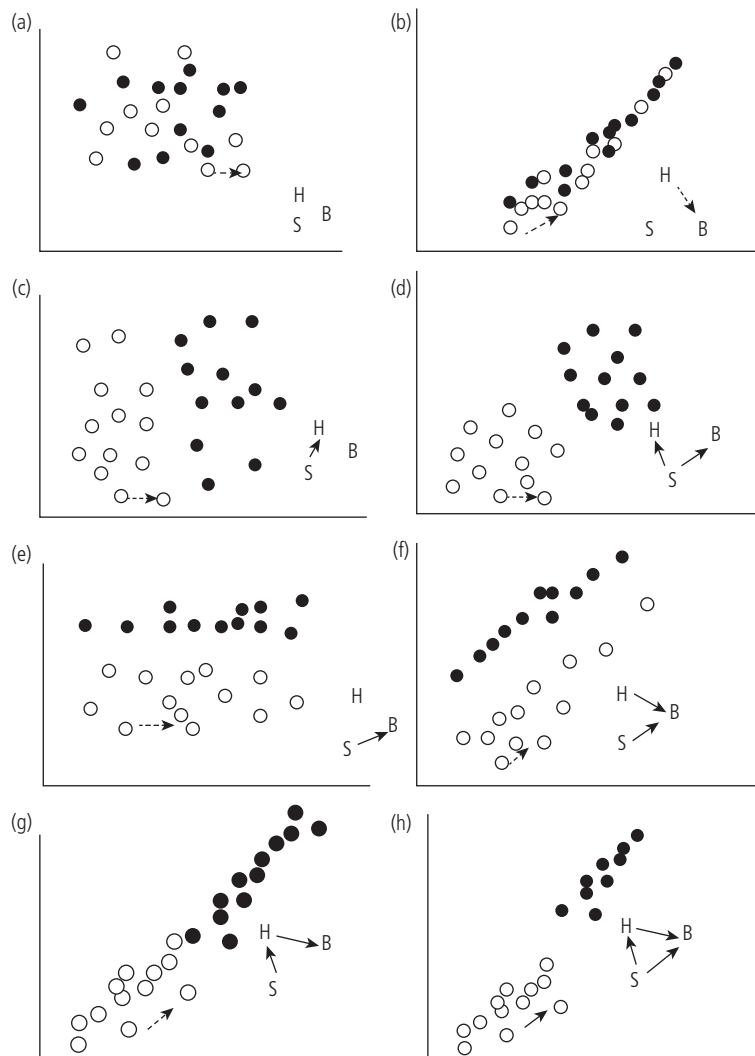


Figure 6.4 Schematic representation of a (not comprehensive) range of idealized scenarios ((a)–(h), see Section 6.5 for details) illustrating how different hypothesized causal links (solid arrows) between habitat (H), system (S) effects, and biodiversity (B) affect expected patterns. Arrows (inset) denote assumed causal links. Dotted arrow illustrates the likely effect of habitat enhancement on biodiversity for a non-organic farm under each scenario. Filled points represent organic farms, open points, non-organic farms.

explained by habitat—only in the final two scenarios is this wholly (g) or partially (h) true. Because the output of statistical models for scenario (g) could be close to those for (d), a full disentangling of the relationship between system, habitat, and biodiversity depends on understanding the *within-system* effects.⁷

The possible links between system, hedgerow, and spiders provide a plausible example of how different statistical outputs are consistent with different patterns of cause and effect (bearing in mind the usual caveat for any associative model that correlation does not indicate causation). Consider the B axis as representing spider diversity while the H axis indicates a metric of increasing hedgerow complexity (it could be botanical or structural, or both). If there is a causal link between hedgerow and spiders, while system is irrelevant, we expect to observe scenario (b). In scenario (c), we visualize a situation where organic farmers have higher quality hedgerows but spider diversity does not respond to that quality. If organic farmers have higher quality hedgerows and, for reasons unconnected with hedgerow (lower pesticide impact, for example), also more diverse spiders, we expect to see pattern (d). A system effect on spiders independent of any hedgerow effect is described in (e). In (f) both system and hedgerow have an effect on spiders, in the absence of any link between system and hedgerow. In the final two scenarios, spider diversity is higher on organic farms *because* organic farms have hedgerows that favour spiders, the final scenario also having an additional direct system effect.

6.5.1 Site selection and sampling methods

For the purposes of this study, we focused on farms that had cereal fields, and we sampled both autumn-sown, referred to as ‘winter’ cereals (commonly grown on non-organic farms), and spring-sown, referred to as ‘spring’ cereals (more often grown on organic farms). Autumn sowing has greatly increased over the past 60 years while spring sowing has decreased, having a number of impacts on the farm environment, notably a significant reduction in overwinter stubbles. The organic farms in the study were: (1) at least 30 ha in area, (2) not highly fragmented holdings (i.e. where organic fields were interspersed with non-organic fields), (3) not predominantly agro-forestry or horticultural,

⁷ These scenarios are not, of course exhaustive; they do not, for example, allow for interaction between predictors. It is possible, for example, for the effect of habitat to be system specific.

and (4) growing the ‘right’ crops in the ‘right’ years (spring cereal in 2000, and winter cereal in 2002 and 2003). Virtually all suitable organic farms in England growing relevant crop types (winter-sown wheat and spring cereals) at the time of the study were examined. The organic farms were paired with non-organic farms using a procedure that was purely geographical and not based on any attributes of either system. The selected study farm pairs were widely distributed, but there was a cluster to the east and south of Bristol (Fig. 6.5).

Plants and invertebrates were sampled at the ‘field’ scale; 89 pairs of cereal fields (‘target fields’) were sampled over three years. Both the spring cereal and the winter cereal fields were approximately equally divided between recently converted (<5 years) and old organic (>5 years). Plants were recorded in plots in the field boundary, the crop edge, and within the field. Spiders and carabid beetles were sampled using pit-fall trapping, before and after harvest. Eighteen traps per field sampled the crop and uncropped boundary habitats, with nine traps in each habitat. All spiders and carabid beetles captured were identified to species



Figure 6.5 Location of pairs of organic and non-organic study farms.

level. Bats (Box 6.1) and birds were studied at the farm scale. Bird surveyors visited the target field monthly between October and February, recording birds seen during one walk around the perimeter of the target field, and one walk across it. Summer bat surveys were completed pre-harvest on 65 farm pairs between June and August in 2002 and 2003. Winter surveys of birds were carried out on 61 farm pairs on the target field and up to five adjacent fields. Birds were mapped on large-scale maps and individual records were subsequently allocated to habitat categories. The focus was on wintering, rather than breeding, birds because it was thought likely that in winter, flocking seed-eating birds in particular might be drawn into organic farms if these farms provided concentrations of seeds in the wider landscape.

6.5.2 How did organic farms differ from non-organic farms?

Our study quantified substantial differences between the two farming systems that might influence biodiversity (Fuller et al. 2005; Norton et al. 2009). Boundary density (km/ha) of all boundaries (including hedgerows) was higher on organic than non-organic farms (see Table 3 in Norton et al. 2009 for details). Hedgerows had fewer gaps and were larger on organic farms. They were also higher and wider than those surrounding non-organic fields. The number and diversity of trees and shrub species in the hedges were similar.

Organic farms had more grassland (as a proportion) compared to non-organic farms and organic target fields were smaller than their non-organic pairs (Norton et al. 2009). Organic and non-organic farms were similar sizes. They also had similar areas of woodland, permanent pasture, and numbers of ponds. There was no evidence that set-aside management differed (organic and non-organic set-aside were equally likely to be rotated or permanent).

Organic farmers tended to sow crops later than non-organic farmers and the crop rotations differed, with organic systems always including a grass ley as part of a cereal/vegetable rotation. Approximately a fifth of non-organic farms cropped continuously (set-aside excluded), but no organic farmers did this. Organic farms were more likely to include livestock (and a wider variety of types) and were more likely to use them on arable land. Organic farmers cut their hedges less often and were more likely to use a traditional hedge management method (laying). More organic farms had agri-environment agreements (in addition to the Organic Farming Scheme) than non-organic (Norton et al. 2009).

6.5.3 The effects of organic farming on diversity and abundance of plants, invertebrates, bats, and birds

Organic farming was mainly associated with positive effects on biodiversity, although there was substantial variation in the size of effects among taxonomic groups (Fuller et al. 2005). Species density and abundance were typically higher on organic farms, but patterns of diversity were less clear (Fuller et al. 2005). The direction of the effects was consistent, with all but one of the significant differences relating to higher diversity or higher abundance on organic farms compared to non-organic farms (Fig. 6.6). The largest and most consistent effects were for plants and the smallest for carabid (ground) beetles. Organic fields were estimated to hold 68–105% more plant species and 74–153% greater abundance of weeds (measured as cover) than non-organic fields, and cover of weeds was consistently higher at all distances into the crop (Fuller et al. 2005).

Chamberlain et al. (2010) looked in more detail at the bird effect illustrated in Figure 6.6. Of 16 species considered, none was more abundant on non-organic farms, and six out of 16 showed statistically significant effects. Variation in habitat abundance of the type detailed by Norton et al. (2009) was thought to be a plausible explanation for the effect, though there were no habitat effects for some species. Hedgerow density, the proportion of arable area at the farm scale (for stock

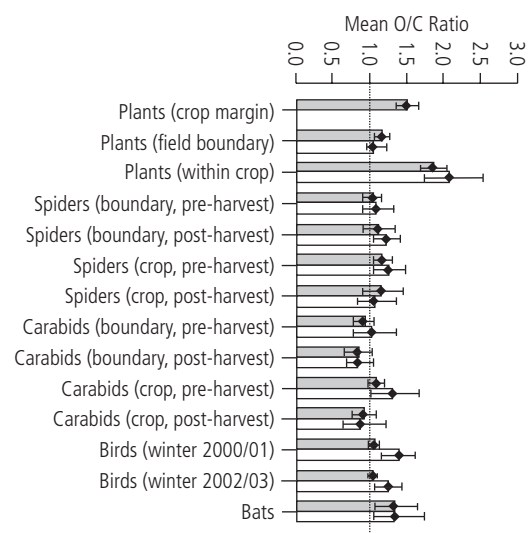


Figure 6.6 Effect of farming system on number of species (grey bar) and abundance (white bar) with confidence intervals. Dotted reference line at ratio = 1.0 indicates no system effect.

Box 6.1 Bat activity on organic farms

Agricultural intensification has been linked with population declines of many bat species. One study of 24 farm pairs suggested bat diversity and abundance were lower on conventional than organic farms (Wickramasinghe et al. 2003), though most of these differences were for water-associated species. Whether any differences relate to 'system' or to non-crop habitat remains unclear. We compared pre-harvest bat activity on 65 organic and non-organic farm pairs in the UK, and explored the effects of both farming system and habitat. Surveyors used methods comparable to those employed by the National Bat Monitoring Programme Field Survey (NBPFs). Activity of *Nyctalus leisleri*, *Nyctalus noctula*, and *Eptesicus serotinus* was identified using heterodyne bat detectors tuned to 25 kHz. Bat passes and feeding calls were counted for each 125 m transect section, at the end of which the detector was returned to 50 kHz and numbers of *Pipistrellus* passes and feeding buzzes were counted for 1 min. Importantly, habitat was sampled in proportion to its availability in the landscape; this meant that any differences in abundance of features such as hedgerows was reflected in the sampling regime. The majority of bat activity was by *Pipistrellus* spp., and for simplicity we therefore consider all bat species together. The rate of encounters was converted to that expected over 3 km, for comparability with the monitoring methodology of the NBPFs.

As with our analyses of spiders and carabids, we compared alternative models and ranked them in terms of their ability to explain the observed data. First, we explored the links between bat abundance and the complexity of the landscape, measured at three different spatial scales, 1 km², 9 km², and 25 km², and asked whether the effect of farming system differed according to landscape type. The amount of land classified as 'open water' was also considered *a priori* as a likely predictor of bat activity. However, almost all values were well below 1% at all spatial scales and this variable may not reflect the availability of water features relevant for bats. Initial screening suggested that models which included this land class could not be distinguished from models excluding it.

As expected, there was strong evidence to link landscape complexity with bat activity: the model that included only farming system, pair ID, and year were supported much less than those that included a landscape variable. The model-averaged parameter estimate for the system variable across this suite of models was biased on the positive side of zero, but included zero, reflecting collinearity between system and landscape attributes and inflated standard errors of parameter estimates.

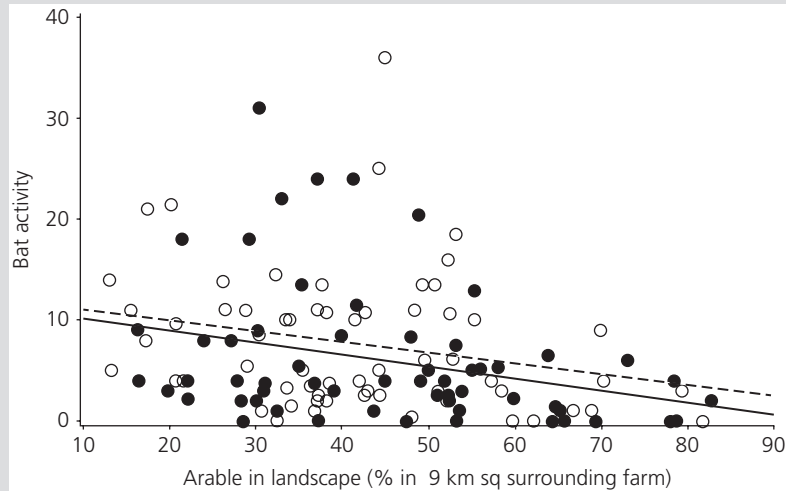
The organic farms tended to be found in landscapes with less arable cropping compared with non-organic farms (a local rather than large-scale effect given the paired design). The mean differences in arable crop abundances (organic minus non-organic) were -7.87% (CI -12.76 to -2.97) at the 1 km² scale, -3.54% (CI -6.68 to -0.41%) at the 9 km² scale, and -1.09% (CI -3.82 to 1.63) at the 25 km² scale. Although models including interactions between landscape complexity and farming system could not be ruled out as implausible, the effect sizes argue against there being any biologically important modification of landscape effects by farming system, or vice versa. The effects of landscape and system are illustrated in Box Fig. 6.1.1.

We went on to explore which aspects of farm structure affected bat activity, in addition to the landscape effect (or whether they were confounded with it, or with system). We also considered if any of the system difference not attributable to landscape could be explained by these variables. The potential predictors included were: **number of ponds on the farm—a binary variable** indicating whether stock was kept on the farm, the proportion of each transect where hedgerow was encountered, the proportion of each transect where water was encountered, the proportion of each transect where stock was encountered, the number of non-cropped habitats present on the farm, the proportion of woodland on the farm overall, and the proportion of pasture on the farm overall.

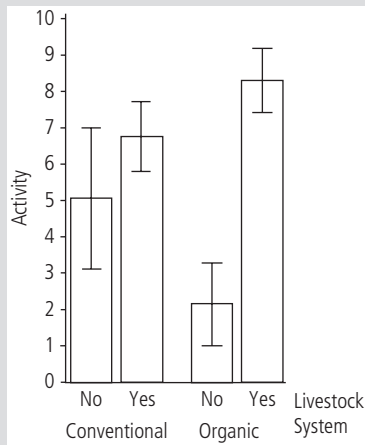
The predictors featuring in the most supported models were ponds, stock, and hedgerow. The presence of stock on the farm is clearly highest weighted by this metric ($w = 0.53$). Those for ponds and hedge were $w = 0.26$ and $w = 0.09$ respectively. Bat activity was higher on farms with stock, and the effect was more marked on organic farms (Box Fig. 6.1.2). The presence of stock was strongly associated with organic systems; while few organic farms surveyed contained no stock (9.2%) a substantial number of non-organic farms were stockless (43.1%). Hence models including the stock effect perform similarly to those with system alone.

The pond and hedge effects may also have been influential. As with livestock, the effect of both these features depended on the farming system. Bat activity increased with the number of ponds on organic farms, but not on non-organic farms, where the slope is negative (Box Fig. 6.1.3); for hedgerows the effect on organic farms was positive, whereas it was consistent with zero on non-organic farms (Box Fig. 6.1.4). Similar results were obtained regardless of whether the outlying farm (number 8) was included in the

continued

Box 6.1 *Continued*

Box Figure 6.1.1 The effect of landscape structure and system on bat activity. Organic farms: open circles, non-organic: solid circles. Least squares regression lines fitted separately for each farm. Dotted line: organic, solid line: non-organic. Bat activity is number of passes expressed per 3 km.



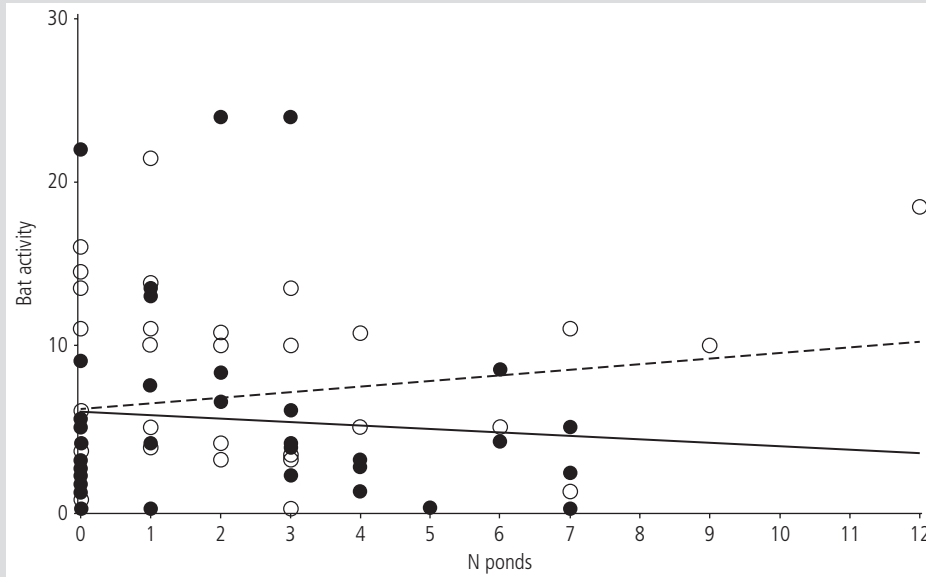
Box Figure 6.1.2 The association between the presence of stock on a farm and mean bat activity (+ /- SE).

analyses. System and landscape were both confounded with both habitats to some extent: organic farms tended to have more ponds than non-organic (mean = 2.70 SE = 0.43) compared with non-organic (mean = 2.50, SE = 0.41), and a higher density of hedgerows (see main text).

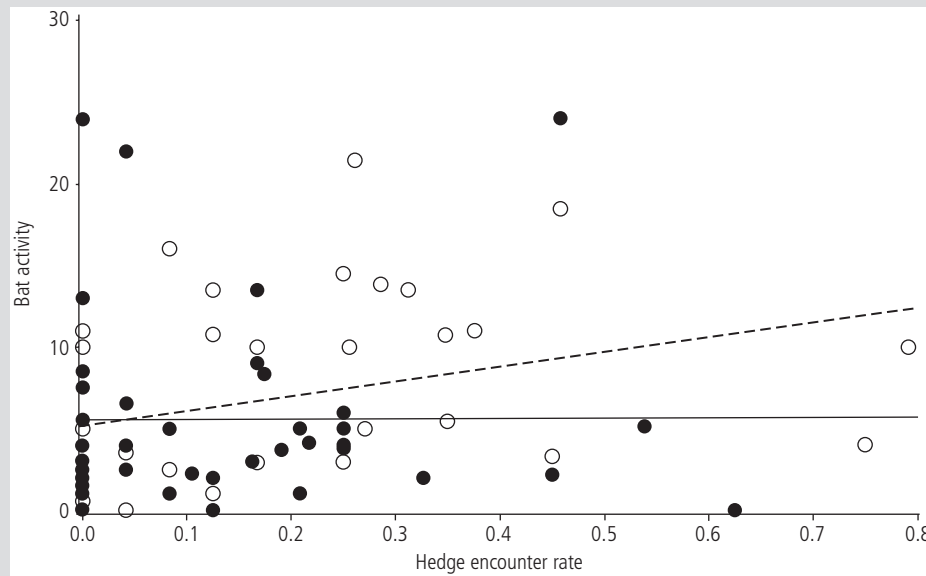
Overall, bat activity levels were low on all farms, and the differences in bat activity seen between non-organic and organic farms were modest. Our work, which sampled habitats in proportion to their availability, shows that much of the British landscape, and particularly that focused on cereal production, therefore offers rather unfavourable habitat for bats regardless of farming system. The mosaic of habitat surrounding the organic and non-organic farms differed, with the broader habitat around organic farms being more complex: some of the apparent benefits we attribute to organic farming could therefore be due to off-farm influences. On-farm features favourable to bats, particularly the presence of livestock, ponds, and hedgerow density were also, to some extent, confounded with farming system, as all were more common on organic than non-organic farms. Nevertheless, our data suggest that the 'quality' of these features in relation to bats was greater on organic than conventional systems. Organic farming therefore does not offer a simple panacea to the decline in bat populations, but the transfer of hedgerow and waterway management techniques to conventional farming, and an overall increase in the number of 'mixed' rather than purely arable enterprises would be likely to yield benefits.

continued

Box 6.1 *Continued*



Box Figure 6.1.3 The association between the presence of ponds on a farm and mean bat activity. Organic farms: open circles, non-organic: solid circles. Dotted line: organic, solid line: non-organic.



Box Figure 6.1.4 The association between transect hedgerow encounter rate and mean bat activity. Organic farms: open circles, non-organic: solid circles. Dotted line: organic, solid line: non-organic.

dove *Columba oenas* and jackdaw *Corvus monedula*), and the grass:arable ratio at the landscape scale (for woodpigeon *Columba palumbus* and jackdaw), were influential.

6.5.4 Local and landscape impacts on invertebrate groups

Here, we investigate in more detail the impacts of farming system and surrounding landscape on two of the taxonomic groups in this study: spiders and ground (carabid) beetles. From results of other studies of the ecology of these two groups, one might predict complex and differing responses to organic farming. Hole et al. (2005) reviewed the evidence for an effect of organic management on biodiversity; they reported considerable evidence for a positive effect on both our target invertebrate groups. However, while the general pattern appeared to be one of higher abundance and diversity in organic fields, it was not universal across studies, and the caveats of small sample size and limited geographical scale remained.

Recent work has shown that landscape context influences the extent to which organic farms have impacts on biodiversity (Winqvist et al. 2012). For example, Schmidt et al. (2005) showed that the species richness of ground-dwelling spiders in crop fields was linked to large-scale landscape complexity irrespective of farming system; more spider species were recorded where the surrounding landscape had a higher proportionate area of non-cropped habitats. They attributed the effect to a higher availability of refuge and over-wintering habitats. The density of spiders responded instead to more local management practices including, in this case, organic farming.

Similar effects apply for carabids. Local conditions such as vegetation and microclimate are important for carabids (Thiele 1977), but carabid assemblages are also strongly influenced by the quantity and arrangement of habitat elements at the scale of the landscape (Burel 1989; Millán de la Peña et al. 2003). Jonason et al. (2013) showed that, for carabid beetles, species richness (and weed seed predation by carabids) was influenced more by wider landscape context than local factors. Other taxonomic groups have shown similar effects: local plant species richness in arable fields is greatly influenced by processes operating at the landscape scale (Gabriel et al. 2005), and similar effects have been observed for arable weeds (Roschewitz et al. 2005). Butterfly abundance and species richness was significantly greater only on organic farms in homogeneous landscapes in Sweden

(Rundlof and Smith 2006; Rundlof et al. 2008), and Holzschuh et al. (2008) demonstrated that an increase in organic farming in the surrounding landscape increased bee species richness and bumblebee density at the local level.

Using data from the study of 89 pairs of organic and non-organic farms, we asked whether we could detect responses of spiders and carabid beetles to organic farming, and whether these differed between the two groups at local and landscape levels. We used data from the two crop types (spring and winter cereals) separately, and data from uncropped field margins and cropped field centres and analysed them separately. Two responses were considered: abundance (the mean number of individuals in traps within the target field) and species richness of spiders and carabids in the trap samples. Spiders can be differentiated into two ecotypes based on their dispersal strategy. Hunting spiders, which include the wolf spiders (Lycosidae) (Fig. 6.2b), generally disperse by walking, and may be affected by local and landscape habitats differently compared with web-builders, which include the linyphiids (Fig. 6.2a). These spiders frequently disperse long distances by ballooning (Topping 1999; Schmidt and Tschardt 2005). Some juvenile non web-builders also balloon, but the ecotypes are sufficiently distinct that we treated them separately.

For each response, we compared a series of candidate models using the model selection procedure of Burnham and Anderson (2002) and Anderson (2008). Relative model performance was assessed using the Akaike Information Criterion (AIC), adjusted for sample size (AICC). Competing models were ranked using Akaike weights. These are interpreted as the probability that the model in question is the 'best' model of the data set, among the series of models under consideration. The process accounts for model uncertainty, and the resulting estimates are 'unconditional'; they do not depend on any single model. Estimates of the relative importance of each predictor variable were provided by summing Akaike weights across all models in which that predictor variable occurred (Burnham and Anderson 2002). Our set of candidate models was based on competing and biologically plausible hypotheses. First we identified the best model from candidate models including system and two aspects of landscape structure. The percentage of arable land was used as a metric of landscape complexity, as for some previous studies. Models also included the percentage of woodland as a candidate predictor, given the likelihood that woodland will provide a diversity of refuge habitat for mobile species. We used the percentage of these land

use categories at two different spatial scales (1 km² and 9 km²) around the focal farm.

Where a system effect was identified, either for responses where a system effect was previously reported (Fuller et al. 2005), or where a system effect was revealed here via its interaction with landscape (and therefore where a system effect may be confined to certain landscape types), we started with the model including system (and landscape where influential) and compared this model with models including habitat variables. For cropped area responses the habitat variable used was plant species diversity, based on previous work suggesting an influence (e.g. Schmidt et al. 2005), with plants providing both habitats and herbivorous prey species for predators. For field margin responses these were margin and hedge metrics. Interaction terms with system were included to allow for the possibility that the habitat effect differed with system.

We identified 131 species of spider from 29 377 individuals from the winter cereal sampling, and 7815 individuals of 77 species of spider from the spring-sown cereals. A total of 107 species of carabid were identified from 62 162 individuals from the winter-sown cereals, and 74 species from 19 313 individuals from the spring-sown cereals. More spiders and carabids were captured within the crop, compared with the uncropped field margin, particularly so for carabids. This was true for both spring and winter cereals and was consistent for both years where winter cereal was sampled. For example, in 2002, an average of 12.9 carabids (SE = 1.4) and 12.1 (SE = 1.10) spiders per trap were captured in the field margin (before harvest) compared with 34.0 (SE = 3.3) carabids and 14.8 (SE = 0.80) spiders in the cropped area. In 2003, the mean capture rates in the margin were 20.5 (SE = 1.90) and 10.4 (SE = 0.92) respectively for carabids and spiders, compared with 35.3 (SE = 2.68) and 14.1 (SE = 1.22) in the cropped area. Trapping rates were between approximately 5–10 times higher before harvest compared with after harvest for both groups. The after-harvest trapping rates in the cropped area in 2002 were 8.54 (SE = 0.74) for carabids and 3.21 (SE = 0.41) for spiders, with rather similar rates observed in 2003.

6.5.4.1 Farming system effect varies with landscape type

We found that spiders were influenced by both farming system and landscape, but the patterns for hunting and web-building spiders were clearly different. For hunting spiders, system and landscape were both

influential. There was very little evidence that the amount of woodland in the landscape was influential, but the amount of arable in the landscape clearly was (this predictor appeared in many of the best models), although its effect was not simple.

Spiders tended to be more abundant and species-rich on organic compared to non-organic farms; the overall system effect was particularly marked for hunting spiders in the cropped area before harvest (Fig. 6.7a), with our analysis suggesting a population average of 77% more individuals and 36% more hunting spider species on organic farms. In both crop and margin samples before harvest, the farming system effect depended on landscape type. The system difference—that is, that more hunting spiders were captured on organic compared to non-organic farms—was much more marked in non-arable landscapes (Fig. 6.7a).

This is clearly visible if we plot the response difference for each pair of farms against the extent of arable in the landscape. Almost all differences were positive for hunting spiders in the cropped area, indicating a positive organic effect, regardless of landscape (Fig. 6.7b). For hunting spiders in boundary samples (both abundance and species richness), the system effect was apparent only in complex landscapes (those comprised of less than approximately 40% arable; there was a clear system–landscape interaction); species richness followed a pattern similar to that of abundance, for hunting spiders (Fig. 6.7c,d).

After harvest, no effects of system or landscape were detected on hunting spiders in the cropped area. In the boundary, however, there were more hunting spiders on organic farms and, again, this effect was more marked in complex landscapes (Fig. 6.7e,f). In other words, there was an overall system effect (hunting spider abundance was higher on organic farms), while an interaction between system and landscape was again attributable to the system effect being more marked in complex landscapes.

In contrast to the hunting spiders, there was no evidence for any effect of system or landscape on web-building spiders in either before or after harvest samples (Table 6.2). The pattern in the cropped area before harvest (Fig. 6.8a,b) was therefore in marked contrast to that for the hunting spiders.

System effects on carabid beetles, where present, were not consistent in the two statistically significant findings (Fig. 6.9). First, there was a system effect on carabid abundance in two different directions. Before harvest, more carabids were captured in the cropped area on organic than on non-organic farms. Second, after harvest, fewer carabids were captured on organic

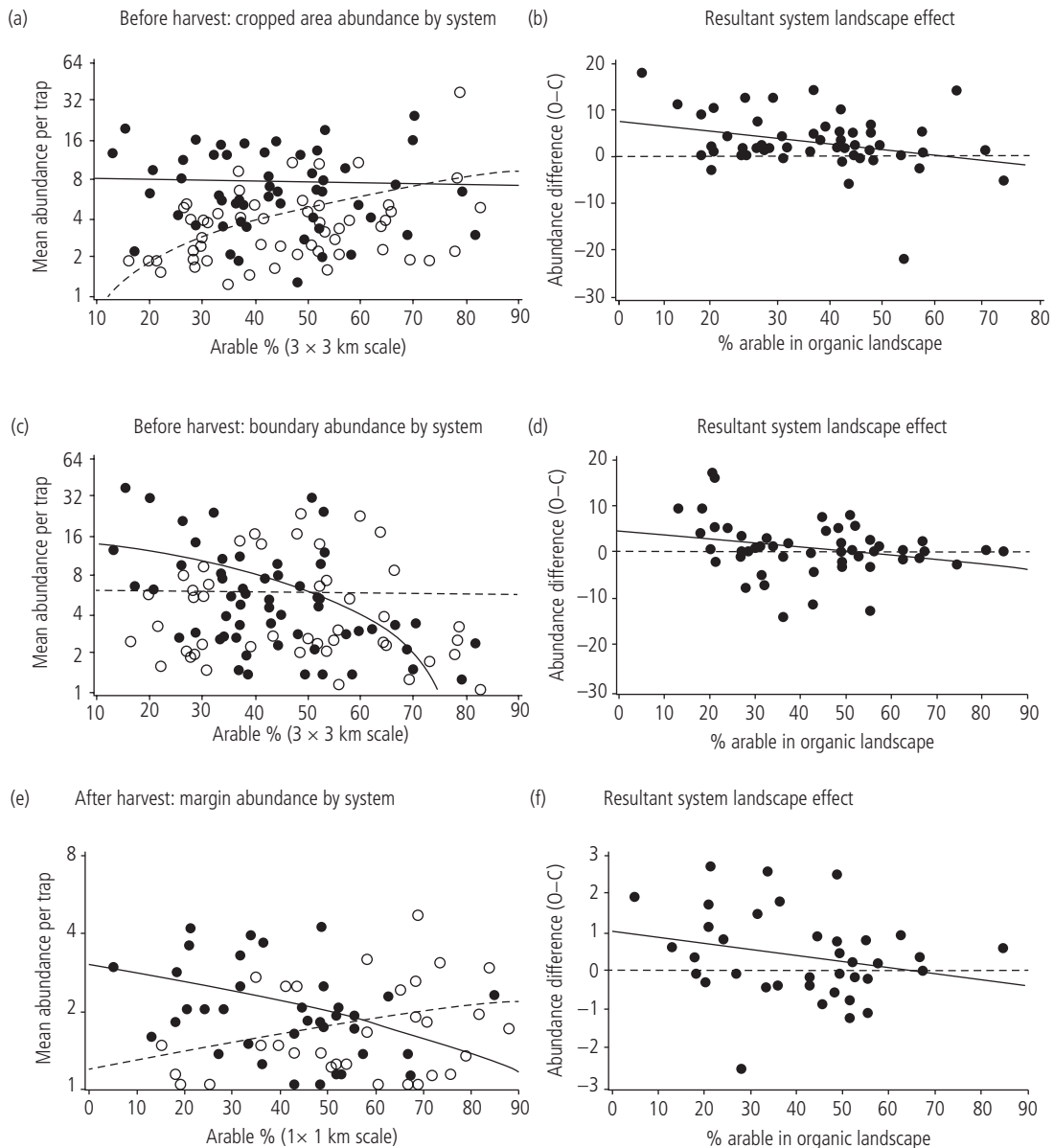


Figure 6.7 Effects of farming system and landscape on abundance of hunting spiders. Organic: solid lines, filled points; non-organic: dotted line, open points. Plots in left-hand panel (a,c,e): lines in plots are linear regressions plotted on log scale by system. Plots in right-hand panel (b,d,f): lines in plots are linear regression (solid line) and reference line (dotted at system difference = zero). Right-hand panel (b,d,f) extracts system effect from left-hand panel (a,c,e).

than non-organic boundaries, though the size of the effect was not large (Table 6.2). For carabids, there was no evidence for any landscape–system interactions, and the observed system effects were therefore not specific to any landscape type.

6.5.4.2 Does habitat explain the system effects?

The clearest system effect occurred for hunting spiders in the cropped area before harvest (Table 6.2). Can we explain this using the field-level habitat predictors? Models including field weed diversity were

124 WILDLIFE CONSERVATION ON FARMLAND: MANAGING FOR NATURE IN LOWLAND FARMS

Table 6.2 Model averaged parameter estimates for system and landscape effects in carabid beetle and spider models. The responses were square root transformed mean species counts per trap, and log mean numbers per trap. Numbers in bold are significant effects.

	Parameter estimate	SE	Effect size	LCI	UCI	Landscape effect	SE
Spiders before harvest							
Cropped area, Hunter abundance	0.578	0.093	1.78	1.49	2.14	0.220 [§]	0.081
Cropped area, Hunter SR	0.31	0.05	1.36	1.24	1.50	0.067 [§]	0.043
Margin Hunter abundance	0.085	0.114	1.09	0.87	1.36	-0.116 [#]	0.108
Margin Hunter SR	0.039	0.066	1.04	0.91	1.18	-0.094 [#]	0.057
Cropped area, Web-builder abundance	-0.066	0.096	0.94	0.78	1.13	none	
Cropped area, Web-builder SR	-0.043	0.056	0.96	0.86	1.07	none	
Margin Web-builder abundance	-0.023	0.088	0.98	0.82	1.16	none	
Margin Web-builder SR	-0.004	0.057	1.00	0.89	1.11	none	
[§] Main effects of arable positive (more spiders and species in arable landscape), but with significant interaction indicating positive trend confined to conventional farms (Fig. 6.7a).							
[#] Significant interaction, indicating negative trends in arable landscapes confined to organic farms (Fig. 6.7e).							
Carabids before harvest							
Cropped area, abundance	0.232	0.12	1.26	1.00	1.60	none	
Cropped area, species richness	0.045	0.059	1.05	0.93	1.17	-0.173[§]	0.035
Boundary, abundance	0.002	0.126	1.00	0.78	1.28	None	
Boundary, species richness	-0.012	0.007	0.99	0.97	1.00	-0.12 [§]	0.004
[§] Fewer species in more arable landscapes.							
Spiders after harvest							
Cropped area, Hunter abundance	0.118	0.143	1.13	0.85	1.49	none	
Cropped area, Hunter SR	0.114	0.071	1.12	0.98	1.29	none	
Margin Hunter abundance	0.165	0.08	1.18	1.01	1.38	0.113 [§]	0.072
Margin Hunter SR	0.156	0.066	1.17	1.03	1.33	-0.18 [§]	0.075
Cropped area, Web-builder abundance	-0.12	0.075	0.89	0.77	1.03	-0.167[#]	0.069
Cropped area, Web-builder SR	-0.097	0.064	0.91	0.80	1.03	-0.115[#]	0.047
Margin Web-builder abundance	0.008	0.08	1.01	0.86	1.18	None	
Margin Web-builder SR	-0.095	0.053	0.91	0.82	1.01	-0.092[#]	0.035
[§] No overall effect of arable, but significant interaction indicating a positive trend on conventional farms.							
[#] Negative trends with increasing arable in landscape.							
Carabids after harvest							
Cropped area, abundance	-0.103	0.145	0.90	0.68	1.20	None	
Cropped area, species richness	-0.066	0.072	0.94	0.81	1.08	none	
Boundary, abundance	-0.254	0.132	0.78	0.60	1.00	none	
Boundary, species richness	-0.177	0.082	0.84	0.71	0.98	none	

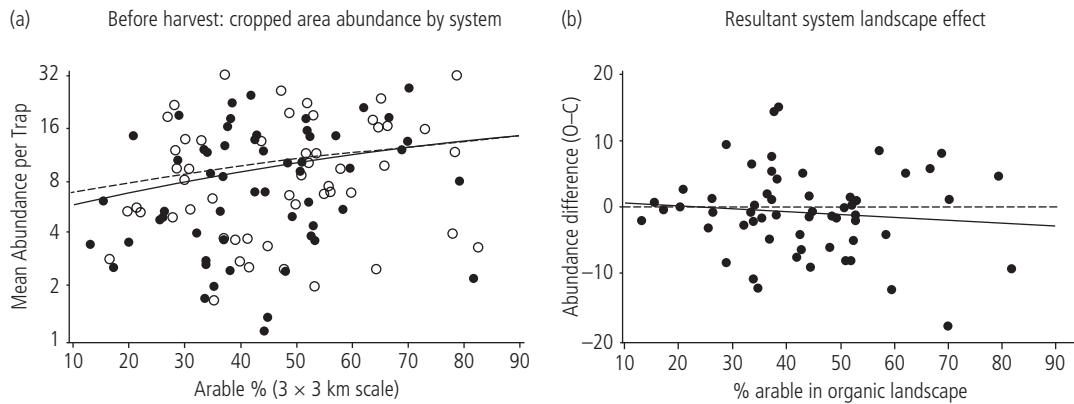


Figure 6.8 Effects of farming system and landscape on abundance of web-building spiders. Organic: solid lines, filled points; non-organic: dotted line, open points. Plot (a): line in plot is linear regression plotted on log scale by system. Plot (b): line in plot is linear regression (solid line) and reference line (dotted at system difference = zero). (b) extracts system effect from (a).

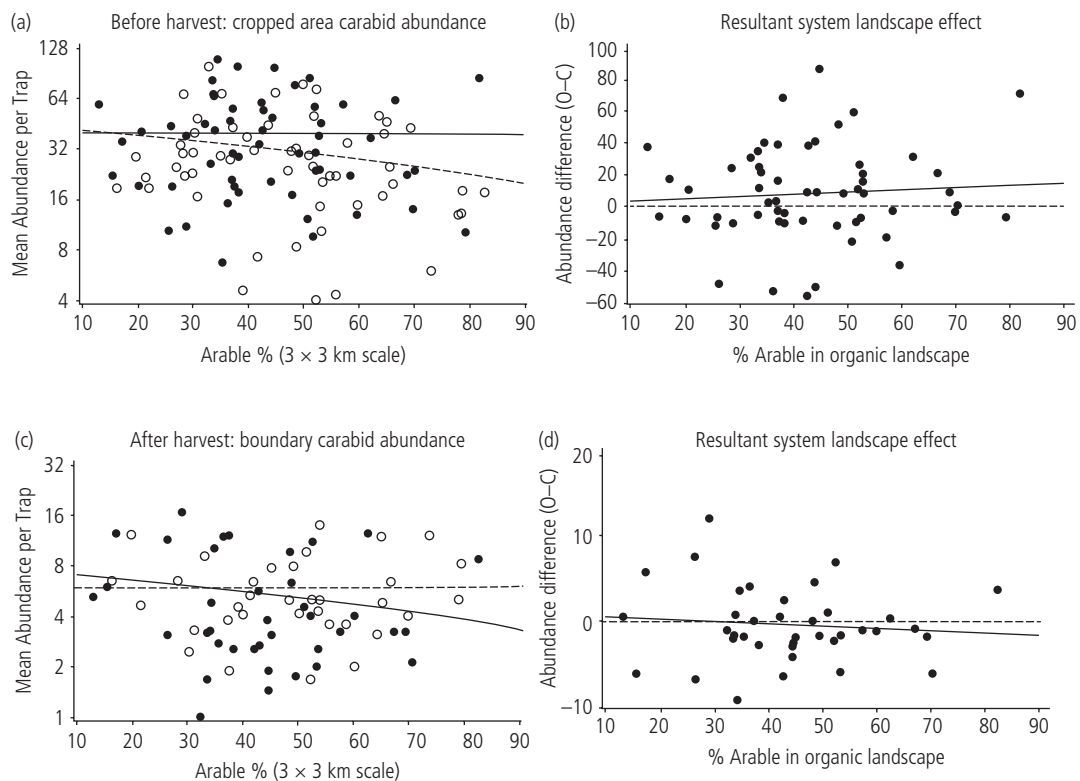


Figure 6.9 Effect of farming system and landscape on abundance of carabid beetles. Organic: solid lines, filled points; non-organic: dotted line, open points. Plots in left-hand panel (a,c): lines in plots are linear regressions plotted on log scale by system. Plots in right-hand panel (b,d): lines in plots are linear regression (solid line) and reference line (dotted at system difference = zero). Right-hand panel (b,d) extracts system effect from left-hand panel (a,c).

ranked higher than the 'baseline' system-arable model and the top models also included interaction between system and weed diversity. In other words, there was some evidence that weed diversity within the crop was influential. Separate modelling for organic and non-organic farms suggested field weed diversity was positively correlated with hunting spider abundance on non-organic farms, but not on organic (the parameter estimates were 0.10 SE = 0.049 and 0.02 (0.01) respectively). System and field weediness are highly correlated, i.e. organic fields are weedy, and the result is consistent with an upward trend where weed diversity is low (on non-organic farms), while no trend occurs over the range observed on organic farms (Fig. 6.10).

The weediness difference does not explain all of the system effect, as the overall system effect remains influential in models which include weed diversity. At the same time, the tendency for the within-pair spider abundance difference to be larger where the weed diversity difference is larger demonstrates that this variable contributes to the system difference, though this relationship is not strong (Fig. 6.11). A similar pattern was observed for hunting spider species richness in the cropped area (Fig. 6.12). There was no evidence that any non-cropped habitat—hedger, for example—had any effect.

There was little evidence from our results that differences in spider abundance and species richness between organic and non-organic farms were driven by differences in non-crop habitat. So the answer to one of our main questions—can the system difference be explained by differences in non-crop habitats?—is 'no'.

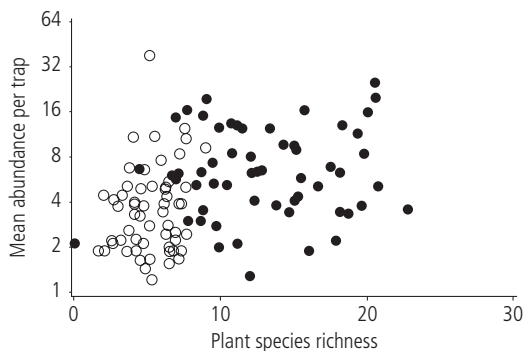


Figure 6.10 Hunting spider abundance and field weed diversity (winter wheat before harvest). Organic farms: solid circles, non-organic farms: open circles. Upward trend in non-organic farms, no trend among organic farms.

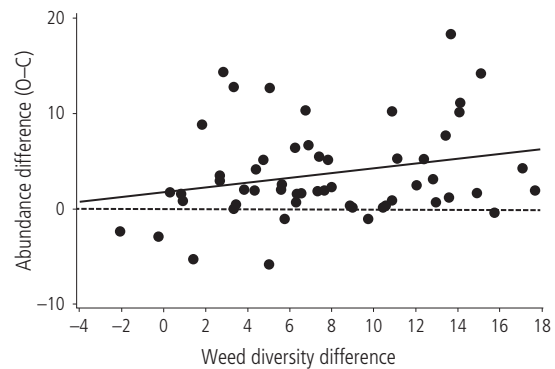


Figure 6.11 Farm pair difference in cropped area: hunting spider abundance and field weed diversity ($r = 0.25$, $P = 0.06$).

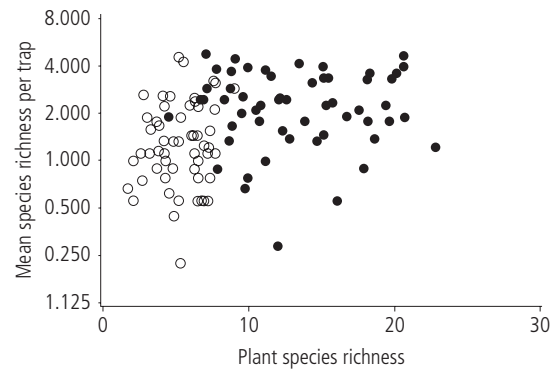


Figure 6.12 Hunting spider species richness and field weed diversity (winter wheat before harvest). Organic farms: solid circles, non-organic farms: open circles. Within system, there is no evidence for any association ($r = 0.16$, $P = 0.22$, $r = 0.07$, $P = 0.61$ for non-organic and organic respectively).

Both spiders and carabid beetles are surface-active predatory invertebrates which are abundant in arable fields, and play a potentially important role in crop pest control of arable crops (e.g. Lang et al. 1999). Their numbers and diversity are likely to indicate those of their prey, especially small invertebrates including mites, Collembola, and aphids. The observations reported here suggested that, for hunting spiders, the between-system difference may be related to weediness within the crop. This is likely to be due to the absence of herbicides in organic fields, resulting in larger weed populations, which provide structural complexity for the spiders.

We were unable to investigate the effects of agrochemical applications on spiders, as these were

entirely confounded with management system. However, reduced abundance of predators with increased use of agrochemicals has been recorded; for example, Greig-Smith et al. (1991) found the densities of linyphiid spiders in areas receiving full pesticide inputs to be approximately 47% of those levels in reduced-input areas. Similar patterns were observed for staphylinid and coccinellid beetles (Vickerman 1992). Spiders and beetles on organic farms should not suffer from direct impacts of pesticide use, although they may be susceptible to non-organic farm practice occurring on neighbouring farms through spray drift or movement of individuals into pesticide treated areas.

Landscape context was influential in our analyses, with the impact of organic farming found to be greater in more complex landscapes. Schmidt et al. (2005) showed that high percentages of non-crop habitats in the landscape increased local species richness of spiders from 12 to 20 species, irrespective of local management and suggest that larger species pools are sustained in complex landscapes, where there is higher availability of refuge and overwintering habitats. Similarly Bergman et al.'s (2004) findings that butterfly assemblages are affected by the surrounding landscape at a large scale led to their conclusions that single-patch management might fail to maintain a diverse butterfly assemblage. Increasing some aspects of landscape complexity might be achieved by increasing the uptake of AES in the surrounding landscape, which can have a positive impact on numbers of moths (Merckx et al. 2009) and numbers of bird species of conservation concern (Dallimer et al. 2010). It is clear that the interactions between invertebrate communities and local and landscape-scale factors are complex; our results for spiders, for example, contrast with those of Rundlof and Smith (2006), who showed that the beneficial effect of organic farming on butterflies was detectable only in intensively farmed homogeneous landscapes. Consideration of spatial scale (from local to landscape/region (Clough et al. 2005) and functional group (Clough et al. 2007)) are important for explaining patterns of invertebrate activity.

Mobility of the species or group concerned is likely to be important. In our study, ~~the generally less mobile guild of~~ hunting spiders ~~was~~ affected more by organic farming and by surrounding landscape composition than web-builders, many species of which can disperse long distances. Dependence on overwintering sites and temporal and spatial closeness to prey populations may be key factors for the less mobile species—increased landscape complexity may result in improved biocontrol by predator populations

(Östman et al. 2001). Our biodiversity data support the conclusions reached by Gabriel et al. (2009), who developed models that show environmental factors associated with lower agricultural potential predispose conversion of farms to organic production, and that these factors naturally create regions with a high prevalence and intensity of organic farms. They argue that the most efficient conservation strategy for English farmland biodiversity would be to encourage the conversion of non-organic farms to organic production in regions where organic farming is already prevalent, and to maximize the intensity of production in areas where it is not. This raises the related question of whether the benefits of organic farming for biodiversity increase disproportionately as more of the landscape is managed organically.

Why did carabids in this study show a less consistent response to organic farming than spiders? At the farm scale, spatio-temporal dynamics of field and boundary overwintering carabid species show considerable variation both within fields and boundaries (Holland et al. 2005). Weedy areas are attractive to carabid beetles as sources of seed and invertebrate food, and conversion to organic farming may facilitate the movement of species more usually confined to field margins into cropped areas (Schröter and Irmeler 2013); conversely, Holland et al. (2009) suggest that it may be possible for some types of vegetation cover to act as sink habitats for predatory invertebrates. Carabids may be particularly sensitive to elements of the landscape and less so to structural diversity at the field scale, using hedgerow networks and permanent elements of the landscape for dispersal and overwintering (Fournier and Loreau 2001). In our study, neighbouring farms were used and pairs within the same landscape were selected. However, the quality of the uncropped habitat did vary between systems, with larger and more sympathetically managed hedgerows occurring on organic farms. Pywell et al. (2005), in their study of beetles and spiders on arable land, concluded that hedgerows, rather than uncropped field margin habitats, provided the highest quality overwintering habitat for invertebrates, including staphylinid and carabid beetles, and spiders. Measures to conserve and enhance hedgerow habitats can be encouraged on both organic and non-organic farms, with important benefits for invertebrate and other farmland biodiversity. The long-term data set of Brooks et al. (2012) on carabid populations shows that, although carabids are in steep decline on their study area, the declines were less severe where hedgerows are managed for conservation. Lastly, our analysis using measures of species

richness and abundance may have masked different impacts of agricultural management on the taxonomic and functional structure of carabid assemblages (Cole et al. 2012).

6.6 Conclusions and applied recommendations

The large sample size and wide range of data collected in our study provided a comprehensive assessment of differences in habitat and biodiversity between organic and non-organic arable farming systems throughout lowland England. We confirmed the findings of our earlier work, and that of others, that organic systems are associated with higher levels of biodiversity across a range of taxa (consistent with the review of Hole et al. (2005)). The most striking differences were for plants, where both species richness and cover of non-crop plants were consistently higher in organic fields (on average there were >80% more species within organic fields). We found that the difference in quality of non-crop habitats within farms, and the effects of surrounding landscape suggest that landscape attributes, non-cropped habitat, and crop management all affected biodiversity in ways that interacted and varied between taxa.

Habitats on organic and non-organic farms differed, and across a range of spatial scales. Organic farms had more grass and more non-crop plants within cereal crops, and more often had livestock. There was more hedgerow per unit area on organic farms, and organic hedges were cut less frequently, and were taller, wider, and had fewer gaps. While it might not be practical for non-organic farms to reduce field sizes (to those of organic farms) or to create more hedges, agri-environment scheme funding is available to support the improvement of hedge quality for both organic and non-organic farmers (Natural England 2009), with likely benefits for a range of taxa.

Management of non-crop habitat is not tied to any farming system. The tendency of organic farms to be more often mixed is more intimately bound up with the farming system. There are many ways in which the presence of stock can increase the potential habitats within a farm, for example through increasing the mix of arable and grassland, and therefore landscape heterogeneity, encouraging dense hedgerows as stock-proof barriers, and the input of dung, which encourages soil invertebrates and the bats and birds that feed upon them.

For spiders and carabids, the observed differences were largely confined to the cropped area (Table 6.2)

and were probably caused by factors associated with crop management, the direct effects of insecticide use being the most obvious candidate. The more diverse non-crop vegetation within the crop on organic farms, resulting from reduced inputs of herbicides and fertilizers, is also highly likely to be influential. There was some evidence for an effect of weed diversity on hunting spider abundance. The pattern for species richness fits that of scenario (d) in our Fig. 6.4, with no link between the habitat and biodiversity. The clearer effect for abundance, compared with species richness, points to insecticides as more likely to be responsible. The weed diversity abundance effect for non-organic farms, and not detectable in organic farms, is also consistent with a pesticide effect (as conventional pesticides are not used on organic farms). Better targeting of pesticide use within non-organic farms, and the possibility of field edge management with reduced pesticide inputs ('conservation headlands') could bring benefits; although little is known of how the scale over which such measures are put into place, and the length of time they are in place, affect the biodiversity benefits that accrue. We were unable to detect any effect of time since conversion on the system effect (Fuller et al. 2005), but scrutiny of community patterns may reveal more subtle side effects.

What are the main policy messages from our work? Our study showed that organic farming does have benefits for biodiversity across a range of taxa, although this study was not designed to consider relative agricultural productivity of the contrasting systems. Exactly which elements of organic farming can be transferred into non-organic systems for biodiversity enhancement remains uncertain. Some beneficial features of organic farms, in particular the quantity and quality of hedgerows, but also other features, such as increased plant diversity within crops (via, for example, conservation headlands), could be enhanced in all farming systems with appropriate support. Increasing the amount of grassland (and moves towards mixed farming systems) would be more difficult to achieve. We also found evidence of interactions among landscape and farming systems in their effects on species, highlighting the importance of developing strategies for managing farmland at the landscape scale for most effective conservation of biodiversity. The total area of organic farms relative to non-organic is small (currently *c.* 2.5% of English farmland is organic). If the benefit of organic farming is greater in some landscapes types, policy makers aiming to encourage conversion might need to consider regional targeting. At the moment, organic

farms are concentrated in the more heterogeneous landscapes of south-western England where our results, for invertebrates at least, indicate the benefit of organic farming to biodiversity are likely to be greatest.

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132 WILDLIFE CONSERVATION ON FARMLAND: MANAGING FOR NATURE IN LOWLAND FARMS

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