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Does organic farming benefit biodiversity?

D.G. Hole [a,*], A.J. Perkins [b], J.D. Wilson [c], I.H. Alexander [d], P.V. Grice [e], A.D. Evans [a]

[a] *RSPB, The Lodge, Sandy, Bedfordshire SG19 2DL, UK*
[b] *RSPB Scotland, 10 Albyn Terrace, Aberdeen AB10 1YP, UK*
[c] *RSPB Scotland, Dunedin House, 25 Ravelston Terrace, Edinburgh EH4 3TP, UK*
[d] *English Nature, Slepe Farm, Arne, Wareham, Dorest BH20 5BN, UK*
[e] *English Nature, Northminster House, Peterborough PE1 1UA, UK*

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Abstract

The intensification and expansion of modern agriculture is amongst the greatest current threats to worldwide biodiversity. Over the last quarter of the 20th century, dramatic declines in both range and abundance of many species associated with farmland have been reported in Europe, leading to growing concern over the sustainability of current intensive farming practices. Purportedly 'sustainable' farming systems such as organic farming are now seen by many as a potential solution to this continued loss of biodiversity and receive substantial support in the form of subsidy payments through EU and national government legislation.

This paper assesses the impacts on biodiversity of organic farming, relative to conventional agriculture, through a review of comparative studies of the two systems, in order to determine whether it can deliver on the biodiversity benefits its proponents claim. It identifies a wide range of taxa, including birds and mammals, invertebrates and arable flora, that benefit from organic management through increases in abundance and/or species richness. It also highlights three broad management practices (prohibition/reduced use of chemical pesticides and inorganic fertilisers; sympathetic management of non-cropped habitats; and preservation of mixed farming) that are largely intrinsic (but not exclusive) to organic farming, and that are particularly beneficial for farmland wildlife.

However, the review also draws attention to four key issues: (1) It remains unclear whether a 'holistic' whole-farm approach (i.e. organic) provides greater benefits to biodiversity than carefully targeted prescriptions applied to relatively small areas of cropped and/or non-cropped habitats within conventional agriculture (i.e. agri-environment schemes); (2) Many comparative studies encounter methodological problems, limiting their ability to draw quantitative conclusions; (3) Our knowledge of the impacts of organic farming in pastoral and upland agriculture is limited; (4) There remains a pressing need for longitudinal, system-level studies in order to address these issues and to fill in the gaps in our knowledge of the impacts of organic farming, before a full appraisal of its potential role in biodiversity conservation in agroecosystems can be made.

Keywords: Organic farming; Biodiversity; Conventional farming; Agri-environment schemes; Comparison

1. Introduction

During the next 50 years, global agricultural expansion threatens to impact worldwide biodiversity on an unprecedented scale that may rival climate change in its significance for the persistence of a panoply of species (Tilman et al., 2001). Predictions of an increase in the human global population to around 9 billion (UN, 2003) could result in a further one billion hectares of natural habitat, primarily in the developing world, being converted to agricultural production, together with a doubling or trebling of nitrogen and phosphorus inputs, a twofold increase in the demand for water and a threefold increase in pesticide usage. This is despite the

* Corresponding author. Present address: Department of Zoology, South Parks Road, Oxford OX1 3PS, UK. Tel.: +44 01865 271 202.
E-mail address: david.hole@linacre.ox.ac.uk (D.G. Hole).

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likelihood of a net withdrawal of land from agriculture in the developed world (Tilman et al., 2001).

Throughout Europe, farmland represents the major land use; in Britain for example, 77% of the land area (18.5 million ha) was under agricultural production in 2001 (DEFRA, 2002). As a consequence, a high proportion of Europe's biodiversity now exists on land dedicated to the production of food, where every effort is made to make as great a proportion of primary production as possible available for human consumption (Krebs et al., 1999). Such intensity of production has resulted in the biological simplification of the farmed environment and the creation of semi-artificial ecosystems that require constant human intervention to regulate their internal function (Altieri, 1999). A wealth of evidence now points to agricultural intensification as the principal cause of the widespread declines in European farmland bird populations (e.g. Donald et al., 2001a; Krebs et al., 1999) and reductions in abundance and diversity of a host of plant and invertebrate taxa (e.g. Donald, 1998; Preston et al., 2002; Wilson et al., 1999) over the past four decades.

Loss of biodiversity on this scale has fuelled the debate over the sustainability of current intensive farming practices, that includes fears over water pollution, soil erosion, landscape quality and food safety. Within Europe these fears have crystallised as growing public, governmental and European Union (EU) support for systems that use less intensive practices, such as organic farming, in the belief that these systems are beneficial for the overall health of the agri-environment (DEFRA, 2002; EU, 2002). As a result, the certified organic and in-conversion area within the EU increased from 0.7 million ha in 1993 to 3.3 million ha in 1999, accounting for 24.1% of global organic land area (EU, 2002). In the UK, the organic market has increased rapidly, displaying growth rates of 30–50% per annum (despite a slowdown to 10.4% in 2003, sales exceeded £1 billion for the first time (Soil Association, 2003)), yet domestic organic producers currently supply only 30% of the market (DEFRA, 2002). The recommendations of a recent UK Government Policy Commission on the Future of Farming and Food (Curry, 2002) are set to provide further financial support to the organic sector, with the justification for offering payments to organic farmers relating to the environmental public goods supplied by organic production methods (DEFRA, 2002). (*Note*: Organic farming is subject to national and international law and within the EU is implemented, labelled, controlled and marketed according to EC Regulation 2092/91).

With such potentially large sums of public money being put into the promotion and support of organic farming, what evidence is there to suggest that it can deliver on the environmental benefits that its proponents claim (e.g. Soil Association, 2000)? In this paper we review studies that have explicitly compared the impacts of organic and conventional farming systems on biodiversity in order to address this question (other key aspects in the debate such as landscape quality and food safety have been considered in detail elsewhere; e.g. Shepherd et al., 2003; Stolze et al., 2000; Unwin et al., 1995). Such a review is timely given recent policy developments in the UK (Curry, 2002; DEFRA, 2002) and Europe (EU, 2002) promoting organic agriculture as one potential solution to the continuing loss of biodiversity in agricultural landscapes.

2. Review of comparative studies

A full literature search was carried out using ISI Web of Science® (1981-present), Biological Abstracts® (1985-present) and the literature cited in the collected papers and reports to cover older publications. Only studies published in English, that have explicitly compared the impacts of both organic and conventional systems on biodiversity are included. (*Note*: 'organic' farming may also be referred to as 'biological' or 'biodynamic' (Lampkin, 2002) – we make no distinction between these terms). The term 'conventional' is widely applied to a range of modern management systems and as such, its exact meaning varies across studies. We take 'conventional' to mean any non-organic farming system, characteristic of a particular farming region where a study took place, and that relies on external inputs to achieve high yields. Organic farming is characterised by the prohibition of a majority of synthetic chemicals in both crop and livestock production (Lampkin, 2002). However, it incorporates a range of other management practices, many of which are uncommonly/exceptionally utilised in conventional systems (Gardner and Brown, 1998). Some of these practices are intrinsic (e.g. avoidance of soluble inorganic fertilisers and synthetic pesticides), whilst others are only encouraged by the standards (e.g. field margin management to promote natural predator populations) (Anon, 2001). Studies that looked solely at the biodiversity impacts of these individual components of management practice are beyond the scope of this review. However, in order to provide context, a synopsis of the principal practices characteristic of (but not exclusive to) organic farming and their likely impacts on biodiversity is given in Appendix A. (*Note*: The term 'holistic' is widely used to describe the management approach utilized in organic farming (e.g. Lampkin, 2002; Soil Association, 2000). We understand 'holistic' to mean the set of principles/regulations enshrined in organic farming that determine standards of husbandry and practice across the whole farming system, in contrast to the application of agri-environment prescriptions for example, where the intent is to target specific elements of the farming system).

The literature search uncovered a total of 76 studies comparing organic and conventional management systems, across a broad range of taxa. In reviews of the ecological literature formal meta-analysis can be a useful statistical technique to provide a measure of the magnitude of the effect size (of the dependent variable) in question (e.g. Fiske et al., 1998; Gates, 2002; Proulx and Mazumder, 1998). In this review however, broad heterogeneity in the measures used to assess biodiversity, both within and across taxa, widely differing methodologies and statistical control between studies, and the small number of comparative studies conducted for many individual taxa, render it inappropriate. We therefore provide a *qualitative* review of the 76 studies, in order to identify whether there is any support for the claim that organic farming delivers significant benefits to biodiversity in comparison to conventional management. Hypotheses cited by authors to explain observed differences between regimes are collated and presented, together with principal findings.

2.1. Flora

Ten studies have compared the non-crop vegetation associated with organic and conventionally managed fields. A further five studies which primarily investigated invertebrate abundance also measured botanical diversity. Two studies compared grassland systems, one investigated field-edge hedgerows, with the remainder focussing on arable and mixed farming systems.

With the exception of Weibull et al. (2003), all of the studies investigating the flora of arable and mixed farming systems recorded higher weed abundance and species richness in fields under organic management, regardless of the arable crop being grown (e.g. mean number of weed species in both margins and cereal fields was more than twice as high under organic management (Frieben and Kopke, 1995); density of non-crop flora in conventional cereal fields was around a third of that in organic fields (Hald, 1999)). These differences were greater for broad-leaved weed species such as Fabaceae, Brassicaceae and Polygonaceae, than grasses, which tended to show less variation between organic and conventional fields (Hald, 1999; Kay and Gregory, 1998; Moreby et al., 1994), suggesting that broad-leaved species are less able to tolerate the intensive weed control measures and denser crop swards of herbicide-treated, heavily fertilised conventional arable fields (Hyvonen et al., 2003; Kay and Gregory, 1998, 1999; Moreby et al., 1994; Rydberg and Milberg, 2000). In several studies, fields under organic management held considerably more rare and/or declining species (according to the relevant national statistics), including red hemp-nettle *Galeopsis angustifolia* (Kay and Gregory, 1999), corn spurrey *Spergula arvensis* (Kay and Gregory, 1998, 1999), cornflower *Centaurea cyanus* (Rydberg and Milberg, 2000) and corn buttercup *Ranunculus arvensis* (Frieben and Kopke, 1995; Kay and Gregory, 1998). A much smaller proportion of rare plants was confined to conventional fields (e.g. *Anisantha diandra* (Kay and Gregory, 1999), as were more nitrophilous species (Hyvonen et al., 2003; Rydberg and Milberg, 2000)). Many of these latter species (e.g. black grass *Alopecurus myosuroides* and cleavers *Galium aparine*) are now considered to be serious agricultural pests.

Weed abundance was higher in field margins than in the mid-field under both organic and conventional regimes, although these differences were generally more pronounced in conventional fields (Frieben and Kopke, 1995; Hald, 1999; Kay and Gregory, 1998, 1999). Exceptionally, abundance of weeds was lower under organic than conventional management (Brooks et al., 1995). In such circumstances, intensive weed control in organic fields using mechanical methods is likely to be responsible (Pullen and Cowell, 1997). Undersowing of crops and the presence of a clover-ryegrass ley within the crop rotation are also designed in part to limit the amount of weed cover in organic systems (Welsh et al., 1999).

Organic farming had less impact on hedge bottom vegetation, with hedges on organic farms displaying significantly higher species diversity than those on conventional farms (total number of hedge bottom species on organic ranging from 24–53 (SE = 1.6); on conventional 20–36 (SE = 1.1) (Aude et al., 2003)). A lack of both herbicide drift and higher immigration rates from the larger weed species pool present in organic fields are likely causal factors. A similar association between high species diversity of hedge bottom flora and extensive farming systems has been highlighted in a number of other studies (e.g. Boutin and Jobin, 1998; French and Cummins, 2001; Hegarty et al., 1994).

Within grassland systems, differences in vegetation composition between organic and conventional sown pastures tended to be less marked (Frieben and Kopke, 1995; Younie and Armstrong, 1995) (although there was some evidence that organic permanent pastures contained more typical grassland species and a greater species richness, particularly of herbaceous flora, than conventional permanent pastures (Frieben and Kopke, 1995)). This is likely to be a result of the length of time required for species diversity to increase in organic pastures following previous intensive management (Frieben and Kopke, 1995; Younie and Armstrong, 1995). Studies indicate that the natural colonization of grassland to form a diverse sward is a slow and unreliable process, regardless of farming regime, especially where rarer species are largely absent from the seedbank and the surrounding landscape (e.g. Berendse et al., 1992; Hutchings and Booth, 1996).

2.2. Soil microbes (bacteria, fungi, nematodes)

Fourteen studies have investigated microbial/nematode communities under organic and conventional systems. Five general studies have included investigations of soil microbes.

Overall, differences in microbial (bacteria and fungi) communities between organic and conventional systems were limited (Foissner, 1992; Girvan et al., 2003; Shannon et al., 2002; Wander et al., 1995; Yeates et al., 1997). However, there was evidence of a general trend towards elevated bacterial (Bossio et al., 1998; Fraser et al., 1988; Gunapala and Scow, 1998; Mader et al., 1995; Scow et al., 1994) and fungal (Fraser et al., 1988; Shannon et al., 2002; Yeates et al., 1997) abundance/activity under organic systems; Fraser et al. (1988) for example reported a microbial biomass 10–26% greater under organic management. Addition of animal (and green) manures on organic farms was cited as the principal factor, providing a significantly greater input of organic carbon, thereby bolstering (in particular) bacterial populations (e.g. Bossio et al., 1998; Fraser et al., 1988; Gunapala and Scow, 1998).

Nematodes vary widely in life-history strategies and fulfil a variety of functions in soil food webs (Bongers and Bongers, 1998). As such they have received significant attention in studies of farming systems. Mirroring the trend in microbial biomass (on which the majority of groups feed), overall nematode abundance tended to be higher in soils under organic management (Foissner, 1992; Neher, 1999; Yeates et al., 1997), but with genus/group specific traits largely dictating community composition (Berkelmans et al., 2003; Neher, 1999; Neher and Olson, 1999). In general, bacterial feeding nematodes were more abundant under organic management, whilst fungal feeding nematodes were more abundant in conventionally managed soils (Berkelmans et al., 2003; Ferris et al., 1996; Freckman and Ettema, 1993; Neher and Olson, 1999; Scow et al., 1994). Whilst a higher bacterial biomass is likely to be responsible for supporting elevated numbers of bacterial feeders (e.g. Ferris et al., 1996; Freckman and Ettema, 1993; Neher and Olson, 1999), the greater abundance of fungal feeders under conventional management contrasts with the apparent trend for reduced fungal abundance and activity under conventional management (Fraser et al., 1988; Shannon et al., 2002; Yeates et al., 1997). Berkelmans et al. (2003) suggest the high level of fungal activity under organic management reported in some studies is an artefact, whilst Ferris et al. (1996) suggest that the high carbon:nitrogen ratio organic materials commonly added to conventional fields are more likely to select for fungal rather than bacterial-dominated decomposition pathways (Beare et al., 1992). Other studies however suggest that additions of organic matter to soil should increase both bacterial and fungal feeding nematodes, whilst decreasing numbers of plant-parasitic species (Freckman, 1988; Griffiths et al., 1994). Whatever the reason for these inconsistencies, the evidence indicates that microbial communities (including nematodes) are likely to be affected by edaphic factors such as soil type and crop type, as much as by farming regime per se (e.g. Berkelmans et al., 2003; Bossio et al., 1998; Foissner, 1992; Girvan et al., 2003; Neher, 1999; Yeates et al., 1997).

A similar situation was reported in grassland systems where Foissner (1992) found no differences in protozoan or nematode populations, whilst Yeates et al. (1997) found little difference in bacterial activity, but greater fungal activity in organic grassland. Tardigrade and nematode (particularly fungal-feeders) populations were also higher under organic management (Yeates et al., 1997).

2.3. Invertebrates

2.3.1. Earthworms

Six studies have specifically compared earthworm abundance and activity between organic and conventional plots. Seven further studies have focussed on a range of taxonomic groups, including earthworms, three in grassland habitats, the remainder arable-only.

Evidence from comparative studies under arable regimes indicated a general trend for higher earthworm abundance under organic management. Brown (1999a) reported higher earthworm abundance (almost twice the density) and species diversity, both in-field and within grass margins, in organic than conventional fields. Gerhardt (1997), Brooks et al. (1995), Liebig and Doran (1999), and Berry and Karlen (1993) similarly reported that organic sites held larger and more active earthworm populations, whilst Pfiffner and Mader (1997) found a higher number of earthworm species, a higher density (up to two times) and more anecic and juvenile earthworms under organic, regardless of crop type within the rotation. Reganold et al. (1993) meanwhile reported densities as high as 175 earthworms m^{-2} in biodynamic soils in comparison to only 21 m^{-2} in conventional. As in the case of soil microbes, such differences are likely to result primarily from the use of farmyard (and green) manures in organic systems which provide an important food resource (Berry and Karlen, 1993; Brooks et al., 1995; Gerhardt, 1997; Pfiffner and Mader, 1997). Prohibition of pesticide use may also benefit anecic and juvenile earthworms, which occur close to the soil surface and so are most at risk of exposure (Pfiffner and Mader, 1997).

In contrast, Foissner (1992) and Nuutinen and Haukka (1990) were unable to find significant differences in earthworm density or biomass between the two systems, whilst Czarnecki and Paprocki (1997) reported a lower abundance of earthworms in organic arable fields. Although the reasons for these differences are unclear, excessive tillage can have a serious negative impact on populations (Berry and Karlen, 1993), even in the presence of high levels of organic matter input and may result in heavily tilled organic fields supporting lower populations than conventional fields (Czarnecki and Paprocki, 1997). In such circumstances grass margins may play an important role by providing structurally stable reservoir habitats that aid rapid within-field recovery of earthworm populations (Brown, 1999a).

In the three grassland studies, Yeates et al. (1997) reported lower earthworm biomass from soils under organic management (less than a third of the mean biomass under conventional), whilst Foissner (1992) and Younie and Armstrong (1995) found no significant or consistent differences between management systems, except for the unique presence of *Allolobophora chlorotica* in conventional fields and a higher proportion of immature *Lumbricus* spp. in organic fields. Whilst this apparent difference in the response of earthworm populations in arable and grassland habitats could be related to the manner in which organic matter is returned to the soil in grassland systems and its consequences for organic matter decay (Curry, 1994), it remains largely unexplained (Younie and Armstrong, 1995).

2.3.2. Butterflies

Only two studies have compared butterfly communities on organic and conventional farms, with conflicting results. Feber et al. (1997) recorded a significantly higher total abundance of butterflies on organic than conventional farms, in both crop-edges and field boundaries and in both years of the study – a direct result of a greater abundance of non-pest species on organic farms (up to twice the mean density found on conventional farms). In contrast, the two major pest species (large white *Pieris brassicae* and small white *P. rapae*) showed no significant difference in abundance between systems. Significantly more non-pest butterflies (more than double the mean abundance) were recorded in the field margins than in the crop in both systems. However, the authors were unable to detect any effect of farming system on the abundance of either pest or non-pest species for a given crop type (although sample size was small). In contrast, Weibull et al. (2000) found no significant difference in single species abundance, species richness or in several diversity indices (e.g. Shannon-Wiener) between farming systems. This apparent inconsistency between studies is almost certainly a result of the rigorous control for variation in rotation and non-crop habitat between farm pairs in the latter study; the presence of grass-clover leys within organic rotations was the principal reason for the significantly higher non-pest butterfly abundance reported by Feber et al. (1997), a habitat that was effectively excluded in the study of Weibull et al. (2000) (see Section 3). A greater abundance and diversity of food plants in organic field boundaries and a lack of spray drift were also cited as potentially beneficial factors (Feber et al., 1997).

2.3.3. Spiders

Three studies have compared spider assemblages. A further seven have recorded spider abundance and diversity as part of wider investigations, all in arable habitats.

Feber et al. (1998) compared surface-active spider communities in wheat fields and found that abundance and species richness were generally greater in organic than conventional fields, but significantly so at only one of three paired sites. Spider communities as a whole differed between the two management systems. This observation was supported by Basedow (1998), who reported a higher diversity and widely differing community structure under organic management, but little difference in abundance. Similarly, Gluck and Ingrisch (1990) reported a more uniform spider fauna in conventionally managed cereal fields, but found that the two most common species (*Oedothorax apicatus* and *Erigone atra*) were present at significantly higher densities. This was at the expense of the majority of less common groups however, which were found at higher densities in organic cereals.

Results from the seven wider studies were largely similar. Booij and Noorlander (1992), Moreby et al. (1994), Reddersen (1997), Pfiffner and Luka (2003) and Pfiffner and Niggli (1996) all reported a higher abundance of spiders under organic management (up to twice as many spiders on organic (Pfiffner and Niggli, 1996)), although differences were not always statistically significant across studies and years. Berry et al. (1996) in contrast reported little difference in spider abundance regardless of management. No differences in species richness were found between farming systems (Booij and Noorlander, 1992; Weibull et al., 2003).

In those studies that reported a higher abundance (and in some instances diversity) of spiders in organic arable fields, richer understorey vegetation, providing greater structural complexity and a more suitable microclimate (as well as supplying prey species with a greater abundance of plant food) was cited as the principal factor (Booij and Noorlander, 1992; Feber et al., 1998). This could explain the prominence of predatory spiders (e.g. lycosids) under organic management (Gluck and Ingrisch, 1990; Pfiffner and Luka, 2003). Relatively small field areas and botanically diverse margins may also be important, allowing foliage-dwelling spiders to re-colonise rapidly after harvest each year (Basedow, 1998; Gluck and Ingrisch, 1990; Haughton et al., 1999).

2.3.4. Beetles

Beetle communities are the most commonly studied animal group in comparisons of farming systems. Eleven studies have focussed on beetle populations, with a further 10 recording abundance and species richness as part of wider studies, all but two in arable habitats.

Twelve studies reported a generally higher abundance, and some evidence for greater species richness, of carabids on organically managed fields (Booij and Noorlander, 1992; Carcamo et al., 1995; Clark, 1999; Dritschilo and Wanner, 1980; Hokkanen and Holopainen, 1986; Irmler, 2003; Kromp, 1989, 1990; O'Sullivan and Gormally, 2002; Pfiffner and Luka, 2003; Pfiffner and Niggli, 1996; Reddersen, 1997), with four studies indicating the reverse (Armstrong, 1995; Moreby et al., 1994; Weibull et al., 2003; Younie and Armstrong, 1995). Andersen and Eltun (2000) and Brooks et al. (1995) meanwhile, reported generally higher activity densities of carabids on organic fields, but lower activity densities of staphylinids. Krooss and Schaefer (1998) also found a lower activity density and lower species richness of staphylinids on organic, the latter supported by Weibull et al. (2003), whilst Booij and Noorlander (1992) found no clear patterns in staphylinid abundance. Pfiffner and Niggli (1996) and Berry et al. (1996) in contrast, reported a significantly higher abundance of staphylinids under organic management. In the single study of dung beetles, Hutton and Giller (2003) reported significantly greater beetle biomass, diversity and species richness on organic farms (on average 38% more species than on conventional).

Evidence suggested that the distribution and abundance of carabids was linked to variation in vegetation structure, with a number of studies reporting a positive correlation between weed species richness/cover and beetle species richness (Carcamo et al., 1995; Kromp, 1990; O'Sullivan and Gormally, 2002; Pfiffner and Niggli, 1996). As with the spider fauna, a stable microclimate resulting from a more heterogeneous crop structure and increased food supply in the form of arable weeds and their associated animal communities, are probable benefits associated with organic management (Kromp, 1989; O'Sullivan and Gormally, 2002; Pfiffner and Luka, 2003). For a specialist group such as dung beetles, the reduced use of veterinary drugs (e.g. ivermectin), diversity of ungulate species and retention of herbaceous field boundaries in organic farming are also likely to have positive impacts on their biodiversity (Hutton and Giller, 2003). The reasons for the generally lower numbers of staphylinids under organic management however are not immediately clear. In the only explanation offered, Andersen and Eltun (2000) suggest that competition with elevated numbers of carabids on organic fields may suppress staphylinid abundance.

However, the majority of studies, regardless of their general findings, reported inconsistencies within the beetle community, with some groups or individual species clearly favouring organic fields and others conventional. For example, of the carabids, *Harpalus* spp., *Bembidion explodens* and *Clivina fossor* consistently favoured organic over conventional fields, whilst *Trechus quadristriatus* generally showed the opposite relationship. Abundance of other species, including *Pterostichus melanarius* and *Bembidion lampros*, two of the most common species found on farmland in Europe (Thiele, 1977), varied between studies and within studies between years or sites. These inconsistencies are likely to reflect the general consensus that the complex influence of site and cultivation specific parameters on beetles can distort expected patterns of species abundance and/or diversity (Kromp, 1990; Moreby et al., 1994; Reddersen, 1997). For instance, variation in tillage practices between organic farms and in pesticide use between conventional farms, may confound any results, since both deep tillage and wide-scale pesticide application can have substantial and unpredictable impacts on beetle communities (e.g. Carcamo et al., 1995; Kromp, 1999). Soil type is likely to be a further confounding factor (Irmler, 2003).

2.3.5. Other arthropods

Ten studies have investigated other (non-coleopteran) arthropod taxa associated with organic and conventional farming regimes, three focussing exclusively on arthropods and seven as part of more general studies.

Overall, the results of all 10 studies suggest that organically managed fields contain a greater abundance and diversity of arthropods than conventionally managed fields (e.g. Berry et al., 1996; Brooks et al., 1995; Letourneau and Goldstein, 2001; Reddersen, 1997). However, there were clear differences in response between taxonomic groups. Whilst aphids and their natural predators tended to be more abundant in conventional fields, where more abundant food resources are provided by heavily fertilised, faster growing crops (Moreby et al., 1994; Reddersen, 1997), groups such as Acari (mites), Formicidae (ants) and Heteroptera (true bugs) tended to show the reverse (Moreby, 1996; Reddersen, 1997; Yeates et al., 1997). Blackburn and Arthur (2001) found no difference in species richness or diversity of centipedes between organic and conventional farms, but reported a significantly higher density on organic, in contrast to Berry et al. (1996) who reported little difference in abundance. Collembola showed few differences between management regimes (Alvarez et al., 2001; Czarnecki and Paprocki, 1997; Yeates et al., 1997), whilst other groups such as Diptera (flies) and Hymenoptera (sawflies, wasps, bees) displayed inconsistencies between studies (Moreby et al., 1994; Reddersen, 1997). Individual species within groups also displayed inconsistencies, with some more abundant under organic, others conventional.

Again, interactions between site and/or cultivation specific parameters were cited as being responsible for the observed variation; collembolan populations for example are reduced by heavy soil disturbance (Heisler, 1991) regardless of farming regime. Sampling technique may also confound the results. Reddersen (1997) found a considerably richer arthropod fauna (both species richness and biomass) in the field margin than in the midfield, but also that the decrease in faunal richness from margin to midfield was greater in conventional fields. Studies that only sampled close to the field edge (e.g. Moreby et al., 1994) may therefore have underestimated differences between conventional and organic regimes.

2.4. Vertebrates

Considerably fewer studies exist comparing vertebrate populations between organic and conventional systems than for smaller-sized taxa. These studies also differ from the latter in that they tend to focus at a larger scale (i.e. whole farms), rather than at the scale of individual fields or experimental plots within fields.

2.4.1. Mammals

Only two strictly comparative studies of organic and conventional farming and their influence on mammalian biodiversity have been carried out. Brown (1999b) found that activity levels of small mammals (wood mouse *Apodemus sylvaticus*, bank vole *Clethrionomys glareolus* and common shrew *Sorex araneus*) were greater in organic than conventional fields, although there was little difference in density. In both farming systems, margins rather than cropped field areas were the preferred habitat. Increased food abundance through sympathetic management of hedgerows and field-edge habitats was cited as the most significant factor benefiting these and a range of other mammalian species occurring on farmland (e.g. brown hare *Lepus europaeus*, common shrew and wood mouse (Flowerdew, 1997; Tapper and Barnes, 1986; Tew et al., 1994)), since many small mammal species are likely to have been affected by the reduction in insect and weed-seed food resources on farmland resulting from intensification (Flowerdew, 1997).

Wickramasinghe et al. (2003) investigated bat activity and species richness using a paired farm (organic/conventional) design. Both total bat activity (all species) and foraging activity were significantly higher on organic farms (by 61% and 84% respectively), suggesting that bats were actively selecting organically managed habitats (farms were paired for habitat type). Whilst no significant difference in species richness was found between farm types (14 of the UK's 16 species were recorded on organic, 11 on conventional), two species, the greater and lesser horseshoe bats (*Rhinolophus ferrumequinum* and *R. hipposideros*) were found only on organic. Greater activity-levels exhibited by bats on organic farms are a likely consequence of the higher overall abundance of insects recorded (including five key insect families identified as being important in bat diet), compared to conventional farms. Superior habitat quality in terms of linear features (in particular tall hedgerows) and riparian bodies on organic farms may both encourage invertebrates and provide better feeding habitat.

2.4.2. Birds

Five major studies have compared bird communities as a whole on organic and conventional farmland. All five assessed bird abundance and/or species richness, primarily during the summer, with one also examining nest density and nesting success.

All five studies reported greater avian abundance and/or species richness on organic than conventional farms, although there was some inter-study variation. In North America, Beecher et al. (2002) and Freemark and Kirk (2001) reported higher species richness and overall abundance (on average 2.0 and 2.6 times greater respectively (Beecher et al., 2002)) on organic than conventional fields, significantly so for several species (including species with regionally declining populations), with a small proportion showing the opposite trend. These differences related to all guilds and included both within-field and field edge census areas. Both nest density and the number of nesting species were found to be significantly higher in organic than conventional fields (Lokemoen and Beiser, 1997). Consistencies between the two European studies included higher densities of skylark *Alauda arvensis*, blackbird *Turdus merula* and greenfinch *Carduelis chloris* on organic sites. Christensen et al. (1996) found 31 species to be significantly more abundant on organic than conventional farms, with only three showing the opposite trend. Of those species showing greater abundance on organic farms, many had declined nationally over the previous two decades (e.g. lapwing *Vanellus vanellus*, linnet *Carduelis cannabina*, corn bunting *Miliaria calandra*). Similarly, Chamberlain et al. (1999) reported consistently higher mean densities of the majority of individual species in both the breeding and non-breeding seasons on organic farms, although these differences were rarely statistically significant (a degree of pseudoreplication was apparent in the Danish study, possibly accounting for the discrepancy in the number of significant differences detected between the two studies). No species was found to be significantly more abundant on conventional than organic farms at any time of the year. Species diversity was significantly higher on organic farms in only one of three breeding seasons during which field data were collected. However, of 18 species, eight showed a significantly higher density on organic field boundaries in at least one season/year.

A further four UK studies examined species-specific differences between organic and conventional farms. Skylark territory density was generally found to be greater in organic rather than conventional fields (more than double within cereal fields), with some evidence that nesting success was also greater in organic fields (Wilson et al., 1997). As part of a wider study, Bradbury et al. (2000) found yellowhammer *Emberiza citrinella* breeding started slightly earlier on organic farms, but found little difference in breeding success or density and concluded that farming regime was not an important factor determining yellowhammer settlement patterns. Morris et al. (2001) however, reported that yellowhammers used organically managed wheat fields significantly more than conventionally managed wheat as a foraging habitat during the breeding season. Moorcroft (2000) detected no significant difference between organic and conventional farming systems in either breeding density or in a range of measures of breeding productivity of linnets.

A greater abundance and diversity of many invertebrate and plant groups, resulting from organic management, was highlighted as the principal reason for the differences in the avian community between farming systems (e.g. Beecher et al., 2002; Christensen et al., 1996; Freemark and Kirk, 2001). This conclusion is in line with a raft of correlational evidence suggesting that intensification/specialisation of arable and grassland systems has reduced the availability of key invertebrate and seed foods for many farmland birds within conventional systems (e.g. Donald, 1998; Evans, 1997; Fuller, 1997; Vickery et al., 2001; Wilson et al., 1999). Such indirect evidence is supported by autecological studies indicating a direct causal link between reduced food abundance and species-specific population decline in the grey partridge *Perdix perdix* (Potts, 1986; Potts and Aebischer, 1991) and the house sparrow *Passer domesticus* (Hole et al., 2002). The lack of any significant impact of farming regime on linnets is likely to be a result of their exploitation of two principal seed resources characteristic of intensive agricultural systems – dandelion *Taraxacum officinale* and oil seed rape *Brassica napus* (a common crop on conventional farms that is rarely grown organically in the UK because of the need for high pesticide and fertiliser inputs) (Moorcroft, 2000).

For many bird species, the retention of traditional mixed farming (i.e. the integration of both crop and livestock components on the same farm) in a majority of organic systems (Shepherd et al., 2003) is likely to be beneficial through the provision of both arable and grassland habitats in close juxtaposition at the between-field/farm scale (Robinson et al., 2001). A wide range of farmland birds are found in greatest abundance in mixed farming landscapes, particularly during the winter (Atkinson et al., 2002).

For individual species other factors were also important. Management practices commonly associated with organic farming such as crop rotations and spring sowing of cereals for example, provide a more suitable landscape matrix and crop density for nesting skylarks, leading to higher territory densities on organic farms (Wilson et al., 1997). Sympathetic hedgerow and field boundary management is also likely to positively influence species such as yellowhammer, linnet and tree sparrow *P. montanus*, by providing better shelter and nesting habitat, as well as increased food resources (Chamberlain and Wilson, 2000; Chamberlain et al., 1999; Freemark and Kirk, 2001).

In summary, despite the difficulty in identifying the cause(s) of the reported differences in biodiversity between organic and conventional systems, a majority of studies drew on similar conclusions, consistent with the predicted or known biodiversity impacts of individual management practices (Appendix A). Three broad practices that are strongly associated with organic farming were identified as being of particular benefit to farmland biodiversity in general: (1) *Prohibition/reduced use of chemical pesticides and inorganic fertilisers* is likely to have a positive impact through the removal of both direct and indirect negative effects on arable plants, invertebrates and vertebrates. (2) *Sympathetic management of non-crop habitats and field margins* can enhance diversity and abundance of arable plants, invertebrates, birds and mammals. (3) *Preservation of mixed farming* is likely to positively impact farmland biodiversity through the provision of greater habitat heterogeneity at a variety of temporal and spatial scales within the landscape.

3. Methodological issues

Any comparison of the impacts of organic and conventional farming systems on biodiversity is likely to be problematic, largely as a result of the complexity of, and interactions between, the range of farming practices that comprise the two systems. The majority of studies seek to minimise apparently extraneous variation, unrelated to farming system (e.g. soil type; differences in community structure resulting from geographic location) with varying degrees of rigour and success. Some studies then go further, attempting to control for variation in crop-type, non-crop habitat or tillage method, either statistically or within a paired field/farm design. Others consider that such variation is part of the overall difference between regimes. The studies reviewed here comprise both extremes of this spectrum, potentially complicating any unbiased assessment.

Five universal problems can be identified: (1) Variation in the definition of organic farming standards between countries (and to a lesser extent between certification bodies within countries) may impact on the validity of a comparison across studies. Whilst adherence to EC Reg. 2092/91 standards will minimise this bias, some variability will still exist (for example, in differing management constraints on non-crop habitats). (2) Disparity between studies in their control for extraneous variation may result in incorrect conclusions being drawn if, for example, observed differences in community structure are caused by landscape characteristics as a result of poor field/farm-pairing, rather than farming regime. (3) Some of the reviewed studies were carried out over a single season/year and may therefore represent stochastic variability in community structure rather than any difference resulting from farming regime. (4) Considerable variation exists in the spatial scale at which comparisons are made, with studies of vertebrate communities tending to be carried out at the farm-scale and studies of invertebrates/plants at the field/plot-scale. Whilst a field-scale study may increase the likelihood of identifying key beneficial management practices, it precludes any accurate assessment of system-level effects. (5) Different studies use different 'measures' of biodiversity (e.g. abundance; density; species richness; breeding success), potentially complicating any comparison. Furthermore, there is a lack of statistical independence between and within some studies, either where multiple taxa are investigated or more importantly, where multiple studies are carried out by the same group and/or at the same site (e.g. five of the 12 taxon-specific studies of soil microbes were carried out at the Sustainable Agriculture Farming Systems (SAFS) project at UC Davis).

In addition, several factors may have resulted in studies underestimating the scale of any benefits of organic farming to biodiversity: (1) When pairing organic with conventional farms, there is likely to have been a tendency to avoid the largest, most intensively managed conventional units, because of the lack of similarly sized organic units for comparison. (2) There may be a time lag in the response of wildlife communities to any benefits generated by a switch from conventional to organic farming (Chamberlain et al., 1999; Irmler, 2003; Younie and Armstrong, 1995). Whilst some authors indicate the length of time since conversion of the organic fields/farms in their study, many either give no information or are comparing fields/farms that converted only a year or so prior to the study commencing. (3) The spatial scale of land management is such that detecting significant effects of organic systems at the field scale may be difficult, particularly for mobile taxa such as birds and butterflies. Furthermore, the lack of any real spatial continuity between organic enterprises results in organic units existing within a 'sea' of conventionally farmed land. Beneficial impacts at the landscape level may therefore only become apparent when organic farming is practised at a landscape scale. (4) By pairing/statistically controlling for the majority of inter-regime variability in management practice, some studies risk excluding the very differences that may be responsible for creating the observed variability in biodiversity in the first place, introducing a bias in favour of the conventional system (Unwin et al., 1995). Weibull et al. (2000) for example, rigorously paired farms for land-use and habitat, thereby excluding the positive impact of clover-leys on butterfly abundance noted by Feber et al. (1997).

One contrary point should also be borne in mind. It is plausible that those farmers who choose to convert to organic status may be pre-disposed to environmentally friendly farming practices in the first place or may farm land that has previously been managed less intensively and is therefore easier to convert successfully to organic. Even as conventional enterprises, biodiversity may have been greater than 'average' on such farms and may have simply remained at that higher level after conversion.

These issues demonstrate the need for more rigorous, standardised investigations into the apparent differences between organic and conventional farming systems. Well-replicated studies are required that follow the development of flora and fauna on organic farms during and after conversion, and compare them directly with geographically local farms that continue with a conventional regime (Greenwood, 2000). Even then, it may prove difficult to accurately assess scale-effects, such as the impact on biodiversity of many closely associated organic units versus a single isolated farm. Similarly, understanding how, indeed if, results from field/plot-scale studies apply at farm, regional or national scales, and the scaling processes involved, is a crucial area for further research. Thus far, rigorous studies addressing these problems are lacking.

4. Discussion

The majority of the 76 studies reviewed in this paper clearly demonstrate that species abundance and/or richness, across a wide-range of taxa, tend to be higher on organic farms than on locally representative conventional farms (Table 1). Of particular importance from a conservation perspective is that many of these differences apply to species known to have experienced declines in range and/or abundance as a consequence of past agricultural intensification, a significant

Table 1
Summary of the effects of organic farming on individual taxon, in comparison to conventional

Taxon	Positive	Negative	Mixed/no difference
Birds	7		2
Mammals	2		
Butterflies	1		1
Spiders	7		3
Earthworms	7	2	4
Beetles	13	5	3
Other arthropods	7	1	2
Plants	13		2
Soil microbes	9		8
Total	66	8	25

(*Note*: total in table > number of studies in review since it includes multi-taxon studies).

number of which are now the subject of direct conservation legislation (e.g. skylark, lapwing, greater and lesser horseshoe bat, corn buttercup *Ranunculus arvensis* and red hemp-nettle are all UK government Biodiversity Action Plan species). These biodiversity benefits are likely to derive from the specific management practices employed within organic systems (Appendix A), which are either absent or only rarely utilized in the majority of conventional systems (e.g. Gardner and Brown, 1998). Several important caveats apply to this generalization however.

First, a minority of studies indicated little or no difference between systems or that conventional systems are beneficial for some species, across a variety of taxa. Inconsistencies between and within studies are perhaps unsurprising given the complexity of the interactions between a large number of environmental variables and between taxonomic groups. However, these inconsistencies also indicate that the benefits to biodiversity of organic farming may vary according to factors such as location, climate, crop-type and species, and are likely to be strongly influenced by the specific management practices adopted. Whilst the complete eradication of weeds for example is not the intention in organic farming systems (Lampkin, 2002), some organic fields may contain almost as few non-crop flora as some conventional fields (Brooks et al., 1995), in part as a result of the development of new, increasingly effective methods of non-chemical weed control (Bond and Grundy, 2001; Pullen and Cowell, 1997). At present, the organic standards generally only *encourage* practices that specifically promote biodiversity (such as field margin management to enhance natural predator populations) rather than require them, despite some certification bodies placing additional conservation requirements on farmers (e.g. Soil Association, 2002). It is also possible that high premiums for organic produce may encourage a minority of farmers, who do not share the values and underlying principles of organic farming, to adopt organic management purely for financial reasons. The extent to which the potential beneficial impacts of organic farming are met on individual farms will therefore be influenced not only by the standards enforced, but also by the attitude and ethical beliefs of the farmer (Greenwood, 2000; Shepherd et al., 2003), and on the economic realities of the marketplace.

Second, this review highlights a number of methodological inconsistencies and concerns inherent in the design of many of these studies that may have resulted in unintentional bias, leading to erroneous conclusions being drawn (including a tendency to underestimate any positive effects of organic farming on biodiversity). This could suggest that even greater biodiversity benefits might have been found had this bias not been present.

Third, the majority of comparative studies have been carried out in arable or mixed systems. As a result, there is a dearth of information pertaining to the impacts of organic agriculture in pastoral systems, despite the extent and importance of grassland landscapes throughout western Europe. In Britain for example, intensification in pastoral regions and the resulting declines in biodiversity have been equally as profound as those in arable areas (e.g. Vickery et al., 2001), yet the potential for organic farming to deliver biodiversity benefits in these systems is far less clear. The impact of organic farming on biodiversity in upland agriculture is similarly poorly understood. All the studies reviewed in this paper moreover refer to terrestrial ecosystems, despite the likelihood that aquatic ecosystems also benefit (Stolze et al., 2000; Unwin et al., 1995); organic farming (by definition) avoids the pollution of waterways by pesticides and soluble inorganic fertilizers and is also likely to lead to, for example, reduced nitrate leaching with consequent benefits for water quality (e.g. Stolze et al., 2000; Unwin et al., 1995 – although see Stolze et al., 2000; Watson and Phillips, 1997).

These caveats highlight a key issue in the organic debate; namely the necessity for further research before any quantitative assessment of the beneficial effects of organic systems on biodiversity in general can be made. To summarise, there is a clear need for: (1) system-level studies that incorporate all components of management practice that may differ between organic and conventional systems (i.e. non-crop habitat management; presence of grass-clover leys in rotation; etc); and (2) longitudinal studies that assess the capacity of organic conversion to reverse previous biodiversity losses caused by intensification.

Notwithstanding these reservations, this review indicates that the biodiversity benefits of organic management are likely to accrue through the provision of a greater quantity/quality of both crop and non-crop habitat than on conventional farms. Three broad management options, largely intrinsic (but not exclusive) to organic farming and that are likely to be particularly beneficial to farmland biodiversity (at least in lowland systems) can be identified: (1) *prohibition/reduced use of chemical pesticides and inorganic fertilisers*; (2) *sympathetic management of non-crop habitats and field margins*; and (3) *preservation of mixed farming*. The question of whether the combination of these practices (and others) combined at the farm-scale within organic systems generates a greater environmental benefit than would be expected simply from the additive effects of the individual practices (i.e. a synergistic response), as proponents of organic farming contend (e.g. Soil Association, 2000), remains untested however. Further research is required to assess whether the 'whole farm' or 'holistic' philosophy underpinning organic systems offers more for biodiversity than the adoption of specific menus of 'key' practices within conventional systems. Indeed, since it is possible (although rarely achieved) for a conventional farm to sustain equivalent levels of biodiversity as those found on organic farms, through the careful adoption of specific management practices, this could suggest that increases in biodiversity are largely a result of identifiable changes in management, rather than any 'whole-farm' effect (Anon, 1999). This view is supported by evidence indicating, for example, that a significant proportion of the enhanced bird abundance on organic farms may be attributed primarily to an increase in the quality and quantity of non-cropped habitats and boundaries (e.g. Chamberlain and Wilson, 2000; Chamberlain et al., 1999; Freemark and Kirk, 2001).

5. Conservation implications

In the UK, agri-environment schemes (AESs) such as Environmentally Sensitive Areas (ESAs), the Countryside Stewardship Scheme (CSS) and the Rural Stewardship Scheme (RSS) (in England and Scotland) and 'Tir Gofal' (in Wales) seek to provide financial incentives to (primarily conventional) farmers to undertake management practices falling broadly into the three categories outlined above (e.g. Ovenden et al., 1998; Vickery et al., 2004). These schemes have proved successful, at least where management options are ecologically and geographically well targeted (e.g. Peach et al., 2001; Swash et al., 2000). However, organic management currently provides a clear advantage over such schemes in that the farm as a whole is subject to the organic standards, rather than the limited areas on a conventional farm that may be exposed to environmental management under AESs. These differences may become less marked in future as AESs become better resourced under the progressive 'greening' of European agricultural policy. At present however, gaps in the provision of habitat quality and quantity for many species still exist (e.g. Vickery et al., 2004). Whilst the proposed new 'entry-level' AES is intended largely to address the 'quantity' issue (Curry, 2002), the success of AES schemes as a whole will be dependent on a balanced uptake between broad-scale, low cost options applied widely across the countryside and higher cost options targeting specific areas to maximize beneficial impacts on priority species and habitats.

This review indicates that organic farming has the potential to help in achieving this balance. The overall increase in homogeneity across the European farming landscape during the latter half of the 20th century has had a profoundly negative impact on farmland biota (Benton et al., 2003; Robinson and Sutherland, 2002). Despite the pressing need for long-term, system-level studies of the biodiversity response to organic management at the landscape scale, the available evidence indicates that organic farming could play a significant role in increasing biodiversity across lowland farmland in Europe. At the same time, continued growth in the organic farming sector is dependent on sustained consumer and legislative support, which in turn will depend largely on the outcome of the debate over the balance between environmental benefits and resource performance (e.g. Goklany, 2002; MacKerron et al., 1999; Stolze et al., 2000; Trewavas, 2001). This debate is ongoing, but its resolution will help set the scene for agri-environment policy within Europe for the foreseeable future.

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Appendix A. Farming practices characteristic of organic systems and their likely impacts on biodiversity

Farming practice	Probable effects on biodiversity
Prohibition/reduced use of chemical pesticides	• Organic systems rely on a variety of practices (e.g. biological control; crop rotation; mechanical weed control) to manage plant and invertebrate pests (Lampkin, 2002) ⇒ avoids direct and indirect effects of pesticides on target and non-target organisms • *Direct effects*: herbicides ⇒ significant factor in the declines of many once common arable flowers in the UK (Cooke and Burn, 1995) and Europe (Andreasen et al., 1996), e.g. corn buttercup *Ranunculus arvensis*, night-flowering catchfly *Silene noctiflora* and prickly poppy *Papaver argemone* (Wilson and Sotherton, 1994); insecticides ⇒ major negative influence on invertebrate communities (Ewald and Aebischer, 1999), including anecic earthworms (Pfiffner and Mader, 1997), butterflies (Cilgi and Jepson, 1995; Feber et al., 1997) and epigaeic arthropods (Clausen, 1990; Kromp, 1989; Pfiffner and Niggli, 1996) • *Indirect effects*: removal of plant food resources and alteration of microclimate ⇒ negative impacts on invertebrate populations (Bell et al., 2002; Feber et al., 1998; Haughton et al., 1999; Kromp, 1989; Pfiffner and Niggli, 1996); reduction in both plant seed food resources and invertebrate abundance significant factor in the declines of a range of farmland bird species (Campbell et al., 1997; Donald et al., 2001b; Wilson et al., 1999); e.g. grey partridge Perdix perdix (Potts, 1986, 1997), yellowhammer *Emberiza citrinella* (Morris, 2002) and likely to have had a negative impact on mammals such as common shrew *Sorex araneus*, woodmouse *Apodemus sylvaticus* and badger *Meles meles* (Flowerdew, 1997)
Prohibition of mineral-based fertilisers	• Organic systems rely on a variety of practices (e.g. animal and green manuring; traditional crop rotations including a grass-clover ley or legume crop) to enhance soil fertility (Lampkin, 2002) ⇒ avoids detrimental impacts on biodiversity resulting from high levels of inorganic fertiliser application (Sotherton and Self, 2000) and consequent high stocking rates • *Effects predominantly indirect*: elevated crop growth rates ⇒ crop out-competes slower-growing arable weeds (Green, 1990); e.g. cornflower *Centaurea cyanus* (Stewart et al., 1994); increase in crop structural density ⇒ alters microclimate at soil level with potentially negative consequences for invertebrate fauna (Hokkanen and Holopainen, 1986; Kromp, 1989, 1990; Pfiffner and Niggli, 1996); limits foraging and nesting opportunities for bird species; e.g. lapwing *Vanellus vanellus*, skylark *Alauda arvensis* and yellow wagtail *Motacilla flava* (Galbraith, 1988; Nelson, 2001; O'Connor and Shrubb, 1986; Wilson et al., 1997)
Mechanical weeding	• Involves the dragging of tines or hoes across the soil surface to remove young weeds (Pullen and Cowell, 1997) • Often less efficient than using herbicides (Krooss and Schaefer, 1998) ⇒ contributes to a greater abundance of non-crop flora in arable fields, indirectly supporting higher densities of arthropods (Kromp, 1989, 1999) • Can be highly effective under certain conditions (Pullen and Cowell, 1997) ⇒ extensive use may lead to the decline of long-lived winter annuals and support of short-lived summer annuals, potentially leading to a more impoverished weed flora (van Elsen, 2000) • May cause high mortality amongst eggs and chicks of ground-nesting bird species (Hansen et al., 2001) unless carefully timed
Farmyard and green manuring	• Animal waste and green manures (i.e. the ploughing in of specific unharvested crops) ⇒ used to replace nitrogen and other elements and to build up soil organic matter content (Lampkin, 2002) • Generally supports a greater abundance of invertebrates that rely on un-degraded plant matter as a food source, e.g. earthworms (Gerhardt, 1997; Pfiffner and Mader, 1997), carabids (Kromp, 1999), and more diverse microbial communities (Fraser et al., 1988) • Can result in insufficient input of nitrogen into organic systems ⇒ leads to poor crop and weed growth, the development of an unfavourable microclimate and a depauperate invertebrate community (Brooks et al., 1995; Krooss and Schaefer, 1998)
Minimum tillage	• Involves the use of discs or tines to disturb the soil surface without physical turning of the soil (Lampkin, 2002) • Avoids detrimental effects of inversion ploughing (physical destruction, dessication, depletion of food and increased exposure to predators (Stoate et al., 2001)) on invertebrate populations; e.g. earthworms (Gerhardt, 1997; Higginbotham et al., 2000); spiders (Haskins and Shaddy, 1986); collembola (Alvarez et al., 2001) and other macrofauna (Krooss and Schaefer, 1998) • May negatively impact carabids ⇒ often found in greater abundance on ploughed fields (Baguette and Hance, 1997) • May modify floral community (McCloskey et al., 1996) ⇒ minimum tillage tends to favour annual weeds (Albrecht and Mattheis, 1998; Cousens and Moss, 1990) whilst perennial broad-leaved weeds are more common under ploughed regimes (Frick and Thomas, 1992; Higginbotham et al., 2000), as a result of variations in seed longevity and species-specific germination patterns • Effects on vertebrates are largely unknown ⇒ some evidence that minimum tillage may benefit bird communities (Lokemoen and Beiser, 1997; McLaughlin and Mineau, 1995)

Appendix A (*continued*)

Farming practice	Probable effects on biodiversity
Intercropping and undersowing	• Both can be used in a rotation to suppress weeds (Baumann et al., 2000) and increase crop yields (Fukai and Trenbath, 1993) • Undersowing increases vegetation structure and heterogeneity ⇒ enhances invertebrate populations; e.g. sawflies (Hymenoptera: Symphyta), carabids and spiders (Helenius et al., 1995; Potts, 1997; Sunderland and Samu, 2000); provides a greater abundance of invertebrate food resources for birds and mammals, e.g. grey partridge (Ewald and Aebischer, 1999; Potts, 1997) and corn bunting (Brickle et al., 2000) • Subsequent over-winter crop stubbles may provide only limited seed accessibility to granivorous birds as a result of a reduction in the area of exposed soil (Moorcroft et al., 2002) • Effects of intercropping on biodiversity are largely unknown ⇒ increase in heterogeneity may favour increased invertebrate diversity; e.g. polyphagous predators (Altieri and Letourneau, 1982; Sunderland and Samu, 2000)
Sensitive field margin/ hedgerow management/ creation of non-crop habitats	• Actively encouraged by organic standards to bolster natural predator populations (e.g. Soil Association, 1999) • Establishment of field margins and beetle banks ⇒ develops and supports larger, more diverse invertebrate communities (de Snoo, 1999; Haysom et al., 1999; Moreby et al., 1994; Thomas et al., 2002); e.g. predatory beetles (Lys and Nentwig, 1994); provides overwintering sites and refuges following harvest (Frieben and Kopke, 1995; Gluck and Ingrisch, 1990); supports a more diverse arable flora (Wilson and Aebischer, 1995); provides important nesting and feeding habitat for birds; e.g. yellowhammer (Bradbury et al., 2000; Morris et al., 2001), grey partridge (Rands, 1985, 1986), whitethroat *Sylvia communis* (Eaton et al., 2002) and a variety of small mammals (Smith et al., 1993) • Positive hedgerow management ⇒ reduced herbicide spray drift (prohibited in organic systems) prevents impoverishment of hedge bottom (Aude et al., 2003; Jobin et al., 1997; Kleijn and Snoeijing, 1997); results in greater floral diversity and increased invertebrate populations (Boatman et al., 1994); greater width and structural diversity is positively associated with abundance and species richness of breeding birds (Green et al., 1994; Hinsley and Bellamy, 2000; Parish et al., 1994, 1995); provides sheltering habitat for mammals; e.g. brown hare *Lepus europaeus* (Tapper and Barnes, 1986) • Hedgerows and other non-crop habitats provide dispersal corridors and islands in otherwise fragmented landscapes ⇒ facilitate dispersal of; e.g. many bird species (Hinsley and Bellamy, 2000), mammals (Fitzgibbon, 1997; Tew et al., 1994) and beetles (Holland and Fahrig, 2000) • Some bird species favour shorter hedgerows; e.g. whitethroat (Eaton et al., 2002) and linnet (Moorcroft, 2000); skylark and lapwing avoid tall boundary structures (O'Brien, 2002; Wilson et al., 1997)
Small field size	• Requirement for stock-proof boundaries in conventional mixed and organic systems is likely to result in smaller average field size than on specialist arable farms (e.g. Chamberlain and Wilson, 2000) • Evidence suggests small fields support greater biodiversity per unit area (principally as a result of a higher percentage of non-crop habitat separating individual fields) ⇒ abundance and diversity of carabids, spiders and arable flora decreases with distance from field margins (Frieben and Kopke, 1995; Hald, 1999; Jmhasly and Nentwig, 1995; Kay and Gregory, 1998, 1999; Kromp, 1999); large fields support less diverse spider communities (Basedow, 1998; Gluck and Ingrisch, 1990); density of brown hares is higher on farms with smaller fields (Tapper and Barnes, 1986)
Spring sown cereals	• Delayed development of spring-sown cereals (in comparison to autumn-sown) produces shorter, less dense crops in early and mid-season ⇒ preferred breeding and foraging habitat for a number of bird species; e.g. skylark (Donald et al., 2001b; Wilson et al., 1997), lapwing (Galbraith, 1988) and corn bunting (Brickle et al., 2000) • Spring sowing frequently results in stubble fields being left over part or all of the winter ⇒ allows spring-germinating annual weeds to set seed and germinate; e.g. cornflower, red hemp-nettle *Galeopsis angustifolia* (Stewart et al., 1994) and corn marigold *Chrysanthemum segetum* (Wilson and Sotherton, 1994); provides a crucial winter food source (i.e. weed seed and spilt grain) for seed-eating birds (Donald and Evans, 1994; Evans, 1997; Wilson et al., 1996); e.g. corn bunting (Brickle et al., 2000), cirl bunting (Evans and Smith, 1994)
Crop rotation	• Involves the planting of a sequence of crops, including a grass ley (often undersown into the previous crop) – used primarily to control weeds and other pests/diseases; also to enhance soil fertility via the inclusion of a legume (e.g. clover in the grass mix) (Lampkin, 2002; Liebman and Dyck, 1993; Stoate, 1996) • Presence of a grass-clover ley ⇒ significantly enhances populations of non-pest butterfly species (Feber et al., 1997); undersowing encourages invertebrate populations (see above) • Increased crop diversity ⇒ may benefit a variety of species that require a structurally diverse crop/habitat mosaic; e.g. skylark (in order to make multiple breeding attempts (Wilson et al., 1997)), lapwing (require adjacent cereal and pasture (Galbraith, 1988; Tucker et al., 1994)), brown hare (graze a variety of crops at different times of the year (Tapper and Barnes, 1986))
Mixed farming	• The occurrence of arable fields in close juxtaposition with pastoral elements is likely to have significant benefits for biodiversity across a range of taxa ⇒ increases habitat heterogeneity at multiple spatial and temporal scales (Robinson et al., 2001; Stoate et al., 2001; Vickery et al., 2001; and see Benton et al., 2003 for a review)

(*Note*: these practices are not exclusive to organic farming and may be utilised within some conventional systems).

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