Background

This book is primarily about the ecology of hedgerows, but we have included ‘field margins’ in the title of the book because to understand what happens in hedgerows, it is necessary to consider the influences of the adjacent land use, and these are much more complex than when Ernie Pollard, Max Hooper and Norman Moore published their seminal work *Hedges* in 1974 (Pollard et al. 1974). Indeed, in many cases in order to understand what is happening at the local level, it is important to consider the wider landscape-scale issues. Since the publication of *Hedges*, compulsory set-aside has come and gone (at least in the EU; Daugbjerg & Swinbank 2011), several classes of pesticide have been developed and been superseded by others (Jeschke et al. 2011), farmland intensification has continued with continuing loss of hedgerows either through removal (Figure 1.1) or lack of management (Carey et al. 2009a), farmland wildlife has significantly declined (Stanton et al. 2018) and responses such as agri-environment schemes have been introduced in an attempt to stem the losses (Anonymous 2009). In addition, the biodiversity and wider value of hedgerows to farming and society have been increasingly recognised through the concept of ecosystem services (Reid et al. 2005). Hedgerows are given protection in a number of countries (Baudry et al. 2000), though such protection may be partial. In the UK, for example, whilst hedgerows are designated as priority habitat (JNCC 2011), they must exhibit specific characteristics to be eligible for protection, and protection is not available for certain types of hedgerow such as those surrounding curtilages (DOE 1997; Marrington 2010).

This chapter briefly introduces readers to the different components of hedgerows and field margins and some important concepts. Succeeding chapters concentrate on particular issues; in some the emphasis is strictly on one component of the system (such as Chapter 3 on the management of hedges or Chapter 6 on softening
the impact of agrochemicals on hedgerows and field margins by modifying their use at crop edges), whilst others consider the ecology of the hedgerow/field margin system as a whole in relation to a particular species or functional group. Unfortunately, whilst we cover a number of species groups, we do not have space to cover them all so, for example, we do not have a chapter on herptiles (amphibians and reptiles) despite field margins probably being important for them (Reading & Jofre 2009; Carthew et al. 2013; Mendenhall et al. 2014), especially riparian margins (Maisonneuve & Rioux 2001; Prosser et al. 2016), although perhaps not always positively (Joly et al. 2001). Our final chapter introduces a rather ignored topic — urban hedgerows. As components of urban green infrastructure, hedges deliver valuable ecosystem services, including being components of sustainable urban drainage and capturing particulate air pollution (Dover 2015), but they are also valuable as urban wildlife habitats (Chapter 14).

What is a hedgerow?

Strictly, ‘hedge’ and ‘hedgerow’ are different terms – the hedge being the woody component of a field boundary, whilst the hedgerow includes the herbaceous component, the bank and ditch – but this distinction is often ignored and the terms used synonymously (Forman & Baudry 1984). Hedgerows can be found throughout the world including North and South America, Africa, Asia, Australia and Europe.
Introduction to hedgerows and field margins

(Burel 1996; Baudry et al. 2000; Truong & Pease 2001; Breen 2017); unfortunately, the term ‘hedge’ has multiple meanings and varies with geographical location, so in the UK it has been used to describe managed or unmanaged shrubby boundaries, stone walls or turf banks (Pollard et al. 1974; Greaves & Marshall 1987); in northern France hedgerows are typically lines of trees, and the landscape as a whole is termed the ‘bocage’ (Baudry et al. 2000); in Spain, olives are often grown in dense parallel lines and explicitly termed ‘hedgerow orchards’ (Gomez-del-Campo et al. 2017). In over 100 countries, including Thailand, Venezuela, Vietnam, Bangladesh and India, vetiver grass (Vetiveria zizanioides) has been planted densely to produce ‘hedgerows’ which are used in soil and water conservation, and in bioengineering and environmental protection projects (Truong & Pease 2001). In this book, unless otherwise stated, the term ‘hedge’ is used to describe linear strips of managed or unmanaged woody vegetation (shrubs and/or lines of trees, also termed woody linear features, or WLF; Maskell et al. 2008), and other boundary structures are given names that are reasonably descriptive of their morphology, e.g. ‘stone wall’. Müller (2013) provides a classification of the field boundaries of 32 European countries that he surveyed; within it he identifies eight general types of woody vegetation (low, medium and high hedges, tree hedges, rows of trees, single trees and/or shrubs, pollard trees and high pruned trees), over 100 different planting styles of hedge and 135 management styles. A good working definition of a hedge is used in the UK Countryside Survey (Maskell et al. 2008), i.e. to be considered a hedge a WLF should have a minimum length of 20 m and be no more than 5 m wide at the base; if composed of a line of trees, it should only be one tree wide. Note that this differs somewhat from the definition used by Baudry et al. (2000) who considered that a linear feature could only qualify as a hedgerow if it was managed in some way. Also, using the Countryside Survey definition, linear features wider than 5 m would be considered to be small shelterbelts; nevertheless such features will probably function in much the same way as narrower structures, and Baudry et al. (2000) note that North American prairie shelterbelts, planted in response to soil erosion that led to the Dust Bowl in the Great Plains (Gardner 2009), are analogous to hedgerows, with many similar features and functions (Guo 2000).

Definitions and components

The nomenclature of hedges and field margins is deceptively simple yet contains a few pitfalls. As a result, we have included in this chapter a diagram of an arable field margin to illustrate the various components (Figure 1.2). It is interesting to track the evolution of diagrammatic content over time, as it reflects our increased knowledge of the ecology of field margins and the increased emphasis on managing them for biodiversity/ecosystem services rather than simply agricultural production. In Greaves and Marshall (1987), field margins were divided into three main areas: the field boundary, the boundary strip and the crop, with the latter two as fairly uncomplicated structures. In Marshall and Moonen (2002), the boundary strip became the ‘field margin strip’, a term that embraced a wide range of potential management
Diagrammatic representation of an arable field margin to show the various components that may be present (after Greaves and Marshall 1987; Marshall and Moonen 2002). The field margin combines the field boundary, such as a hedgerow or green lane, AND any associated boundary strip. Ditches may be located on both sides of a field boundary (and would be considered to be part of the boundary) and also found either side of the track inside a green lane. WR = wheel ruts typically made by farm vehicles. The field boundary in the diagram is a hedgerow (i.e. the hedge [shrubs/trees] + associated raised vegetated bank); other common boundary features include dry stone walls, grass banks (baulks) or strips, windbreaks, and wood edges. The boundary strip is typically composed of a sown grass strip, buffer or other agri-environment strip often sown with a mix of species e.g. for pollinators or for game cover. ‘ss’ = sterile strip: an area of bare ground created either by use of a broad-spectrum herbicide or by mechanical means (rotovation). The crop margin/headland may be modified by changing agrochemical inputs to allow the growth of broad-leaved plants (whilst controlling most grass species) within the crop to create ‘Conservation Headlands’. Options, and the crop now contained the term ‘conservation headland’ at its edge. In Figure 1.2, whilst I have not included a farm track in the boundary or field margin strip (it can still be present), I have added a green lane and separated the boundary/field margin strip from the crop with a sterile strip. Whilst we identify different components of hedgerows and field margins, it is probably worth remembering that whilst the different components have their own individual attributes, it is their collective value which is most important. Wolton et al. (2013) examined the relationship of 107 species considered as priority and farmland indicator species...
associated with hedgerows with five elements of hedgerows (trees, shrubs, hedge base, field margin and ditches). They found that 65% of species were dependent on more than one hedge component, and 35% were dependent on three or more.

**Crop influences**

The type of crop and its management have a profound impact on the flora and fauna of field boundaries. The major and most obvious division of crops is into grassland and arable. In the latter, hedgerow/field boundary removal to improve efficiency of crop production has led to some areas being largely denuded of semi-natural landscape features and others left with a much reduced stock, and the use of synthetic fertiliser and pesticides has drastically modified field boundary animal and plant communities (Robinson & Sutherland 2002; Petit et al. 2003). In grassland systems, the main division is into grassland that is used for grazing and that used for primarily for grass production (hay, silage, zero-grazing) although late-season ‘after-math’ grazing or early spring grazing may still occur. Field boundary vegetation in permanent, extensive pasture is undoubtedly the most valuable for biodiversity, provided stocking densities are reasonable. Whilst traditional hedge management (laying) was designed to make hedges stockproof, the expense of maintaining hedges in this way has led to the wholesale use of fences. Sheep, in particular, are very good at denuding basal hedgerow vegetation and eroding hedge banks if not controlled. Unfortunately, in some cases, farmers have been known to spray out boundary vegetation around hedges to prevent shorting of poorly sited and poorly maintained electric fences. Protecting the herbaceous field boundary vegetation and palatable shrubs in field boundaries from being completely eaten-out for biodiversity conservation purposes requires protective measures (Figure 1.3). Field boundaries around intensive grassland can suffer from both uncontrolled stock grazing and some of the issues prevalent on arable land; for example, chemical sprays and fertilisers used to manage monoculture grassland can drift into the boundary vegetation, and close ploughing, when re-establishing the sward, can sever shrub and tree roots. In some areas, stock are now maintained in very large open fields. Here, paddocks are not defined by hedges or other field boundaries but are purely notional and defined using electric fencing (Teixeira et al. 2017).

**The field boundary**

Greaves and Marshall (1987) defined a field boundary as being composed of a barrier such as a ‘hedge, fence or wall, the hedgebank if present with its herbaceous vegetation, and any associated water course such as a ditch or drain’ which some e.g. Hahn et al. (2014) appear to have misinterpreted as excluding grassy margins. This latter is understandable given the written description used by Greaves and Marshall (1987) does indeed seem to omit grass strips/banks as a field boundary on their own (i.e. without a hedge/fence/wall), but scrutiny of the figure in the original paper shows that ‘grass baulk’ is included in the ‘barrier’ component. Whilst not
the most accessible term, ‘baulk’ does mean ‘a ridge left unploughed between furrows’ (Sykes 1982), an ‘unploughed ridge’ (Schwartz et al. 1988) or ‘a narrow strip of grass in a common arable field. Used for access, grazing, etc.’ (Hollowell 2000). So the field boundary, whatever it is made up of, is the ultimate structure that delimits a field. Of course, even this is a simplification, as any given field boundary may be made up of a number of these structures in combination, e.g. there may be a length of tall hedge, followed by a section of short hedge with scattered trees, followed by a section of wood edge. The hedged sections may be on a raised bank with a ditch that runs only partly along its length before petering out. Ditches are associated structures which are usually considered part of the field boundary.

**Green lanes**

Green lanes (see Figure 10.3) are unmetalled trackways with field boundaries either side (Dover & Sparks 2001), but the actual configuration (width, length, depth, vegetation) can vary widely due to their organic development from a range of origins, including ancient drove roads (Belsey 1998). Whilst perhaps the ‘classic’ green lane of lowlands has hedges either side (Dover et al. 2000), some may not have hedges at all but be composed of a sunken track (holloways; Muir 1981) such that the

FIGURE 1.3 In intensive grazing systems it may be necessary to take protective measures to prevent the base of hedgerows from being completely denuded.

*Source:* © John W. Dover
earth banks and herbaceous vegetation create the same effect. In upland landscapes, hedges are usually replaced by dry stone walls and, here, green lanes composed of double stone walls can frequently be found. The topography and structure of any given green lane can vary substantially, and it will typically link to many other field boundaries along its length. The most important features of a green lane, from an ecological standpoint, are probably the unsealed track surface, the enhanced microclimate due to wind shelter between the hedges/walls/earth banks, the distinct plant communities they hold, their structure (including presence of trees) and the less intensive management they seem to get compared to other field boundary types (Dover & Sparks 2001; Walker et al. 2006).

**Intersections or nodes**

The term ‘node’ or ‘intersection’ is used in landscape ecology (Forman & Godron 1986; Noss & Harris 1986) in relation to field boundaries where they meet or intersect or where they have junctions with other semi-natural elements such as woods, riparian areas or uncropped patches of land. Such areas often have greater abundance or species richness of flora and fauna than linear sections of field boundaries (Figure 1.4), but this may depend on the nature of the field boundary and management (Lack 1988; Fry 1991; Dover 1996; Dover et al. 1998).

**FIGURE 1.4** A field corner, or ‘node’, in a barley field with abundant wildflowers and arable plants. The crop behind the fenced and ‘gapped-up’ hedge is Jerusalem artichoke, used as a game cover crop. A narrow sterile strip, created by chemical means, clearly delineates the crop headland from the field boundary.

(Source: © John W. Dover)
Boundary strips

Because of the crisis in farmland biodiversity and the increasing recognition of the ecosystem services (especially pollination) delivered by hedgerows and field margins (see below), agri-environment schemes have been developed in an attempt to mitigate some of the impacts of modern intensive farming. One of the most popular approaches is to take out of production a narrow strip (typically between 2 m to 12 m, but in some cases up to 24 m, wide; Haaland et al. 2011) around the field boundary and modify it in some way (see Figure 7.5 and Figure 13.5) or to change the management of the crop margin. In the case of the latter, Conservation Headlands have been developed for arable (primarily cereals; see below and Chapter 4), and whilst the term has also been adopted by some for modifying grass margins (Haysom et al. 1999, 2000, 2004; Cole et al. 2007), the approach is fundamentally different and is described, briefly, below.

There has been much less research aimed at softening the impact of crop management on field margin flora and fauna in grassland systems than in arable (Fritch et al. 2017). However, some studies have been carried out using methods such as simply excluding grazing from the margins of an existing sown sward (sometimes coupled with reduced agrochemical inputs). Other approaches include rotovation or spraying a marginal strip of an existing sward followed by natural regeneration from the seedbed or rotovation followed by sowing wildflowers (sometimes with a grass mix) or other species (Haysom et al. 1999, 2000, 2004; Cole et al. 2007; Sheridan et al. 2008; Potts et al. 2009; Fritch et al. 2011; Anderson et al. 2013; HUallachán et al. 2014; Woodcock et al. 2014; Wiggers et al. 2016; Fritch et al. 2017). Stock control, through fencing, and an annual hay (removal) cut or autumn/winter grazing was still used in these studies. In general, all these approaches tended to have some benefit in diversifying the sward structure/composition and increasing invertebrate abundance/richness even by simply relaxing the grazing management. However, natural regeneration tended to allow the development of a sward with aggressive weeds and is probably not acceptable in most circumstances, so sowing wildflower/grass mixes is likely to have the most benefit even though it is a more costly option.

In arable systems, the exact approaches vary from country to country and with crop type. Haaland et al. (2011) reviewed the literature on the impact of sown wildflower strips (Figure 7.5) in central and northern Europe on farmland insect abundance and diversity. They found that sowing wildflowers around field boundaries, after cultivation, was generally superior to sowing a mixture of grasses or allowing natural regeneration of the seedbed. However, seed mixtures targeted specifically at pollinators or aimed at producing nectar-rich swards could be superior to general wildflower mixes. Species groups responded differentially to strip characteristics (geographical location, seed mixture, sward structure and management, age and flower density), and most species identified as benefitting from margin management in the studies were relatively common. Studies published since the review of Haaland et al. (2011; Appendix 1.1) suggest that the effects of margin strips on biodiversity may be context dependent, i.e. their value appears to be greatest in
homogeneous rather than heterogeneous landscapes (the latter having a higher proportion of semi-natural vegetation than the former) and that, with suitably tailored seed mixtures, they may be a source of natural enemies whose action can potentially reduce the need for pesticide application on the adjacent crop by preventing pest outbreaks reaching the economic threshold for spraying (Tschumi et al. 2015; Appendix 1.1).

**Sterile strips**

Sterile strips (Figure 1.4) are used in arable systems to reduce ingress of weeds into the crop from the field boundary (Vickery et al. 2002). If there is no boundary strip, then the sterile strip will be immediately adjacent to the field boundary, whereas if some kind of boundary strip has been planted it will be located between the boundary strip and the crop edge. Sterile strips can be created in one of two ways, by maintaining a weed-free soil surface by rotovation (i.e. mechanical means) or by spraying a persistent wide-spectrum herbicide. Mechanically managed sterile strips are usually much wider than those created by spraying (due to the need to get a tractor round the crop edge). In the past, the chemical Atrazine was used to create sterile strips (Boatman & Wilson 1988), but this chemical has now been banned in the EU (Sass & Colangelo 2006), and other chemicals, such as Glyphosate, are used instead (Anonymous 2008b).

**Conservation Headlands**

The headland, in agricultural parlance, is the area of a field that is used for turning by agricultural machinery (Muir 1981). The term ‘Conservation Headland’ was adopted by the Cereals and Gamebirds Research Project run by the Game Conservancy Trust (now the Game and Wildlife Conservation Trust) to describe the practice of modifying the management at the edge of cereal crops (Sotherton et al. 1989). Essentially, broad-spectrum herbicides (which kill broadleaved plants and grasses), fungicides with known insecticidal activity and insecticides were not sprayed on the outer 6 m of cereal fields, whilst grass weed herbicides and fungicides shown to have little or no insecticidal activity were. There were modifications to this general rule, e.g. spot spraying of the herbicide fluroxypyr was allowed to control infestations of cleavers *Galium aparine* (Boatman et al. 1988). Hence, ‘Conservation Headlands’, described in detail in Chapter 6, contain a wide range of broadleaved plants, including rare arable ‘weeds’ (Wilson & King 2003) if present in the seed-bank (see Chapter 2), whilst minimising losses to crop yield. A subsequent modification of the technique uses reduced fertiliser inputs (Kleijn & vanderVoort 1997; Walker et al. 2007), which, by reducing crop vigour, allows more light to penetrate the canopy and reduces competition with broadleaved plants. The system was developed initially as a way of improving the availability of invertebrate food for grey partridge *Perdix perdix* chicks during a critical stage of their development but was soon shown to have a wider biodiversity value.
Baudry et al. (2000) give a good roundup of historical studies and cite Rackham (1986), who indicated that archaeological evidence confirms the presence of hedges in Roman times and who also suggested some remaining examples of hedged field boundaries in the Land’s End Peninsula of Cornwall (UK) were of prehistoric origin. Pollard et al. (1974) considered the first reference to a British hedge to be 547 AD in the Anglo-Saxon Chronicles; nevertheless, they considered some field boundaries, if not the hedges on them, to be Roman. More recently, Müller (2013) gives a chronology of field boundaries, suggesting by inference the early Palaeolithic to be the earliest use of dead hedges (i.e. linear barriers composed of cut branches), though probably used as defence against predators, and around 11,000 BC the use of woody/shrubby hedges.

The origin of hedgerows is a subject that continues to fascinate people interested in landscape development and ecology, whether from an amateur or professional perspective. Possibly the most influential work in this respect in the UK was that of Max Hooper, who developed a rule-of-thumb method whereby hedges could be dated by counting the number of shrubs in 30-yard lengths. His work suggested that a hedge's age could be approximately assessed by assuming that one species was added for each 100 years of its existence (Hooper 1970). This method is surprisingly robust provided precautionary measures are taken – including pilot studies correlated with documentary records (Pollard et al. 1974). This caution is necessary, because the number of species in a hedge at any given time is the sum of the number of species when the hedge was first created and the accumulated number of additional species gained over time, as well as the impact of past management, local environmental factors, geology and geography (Wilmot 1980). One of the critical points, then, is the number of species that were in the hedge initially – and this will be strongly related to how and why the hedge was created in the first place. Forman and Baudry (1984) recognised three main origins of hedgerows:

- **Planted** – deliberately created, typically using a single species (although multispecies plantings may have arisen from whips derived from nearby woodland (Pollard et al. 1974)). Plantings may be on a bank and associated with a ditch.
- **Spontaneous** – trees and shrubs develop along pre-existing structures such as unhedged field boundaries, fences, walls or ditches from seeds dispersed by wind or animals such as birds.
- **Remnant** – typically a result of tree clearance where a strip of trees/shrubs is left along the ownership boundary.

The earliest field boundaries were probably simple structures made with whatever material was at hand, developing in sophistication over time. Whilst spontaneous hedges require an influx of seeds to initially develop, both planted and spontaneous hedges require seed dispersal to increase in species richness over time (as per Hooper’s rule). Forman and Baudry (1984) examined the origin of shrubs and trees
in two European and two North American sites and concluded that, in the majority of cases, hedge shrubs were bird dispersed, whilst trees were primarily wind dispersed. They concluded that spontaneously developed hedges were primarily composed of bird-dispersed seeds (from a study in New Jersey, USA), whilst the shelterbelt hedges in the Great Plains (USA) were primarily composed of wind-dispersed material, which were probably derived from planted stock. In Europe, the UK hedgerow shrubs were primarily bird-dispersed, but the trees originated from wind-dispersed propagules; in Brittany, the trees were primarily bird or mammal dispersed whilst the shrubs were a mix of wind and bird dispersed.

In the UK, older field boundaries tend to be curvilinear or irregularly shaped, probably as a result of assarting (forest clearance) or following natural boundaries, contours or established tracks (such as drove roads). From the mid-1700s onwards, new field boundaries were often more rectilinear in form resulting from the Private or General Parliamentary Enclosure Acts, which enabled the division of commonly held land into discrete fields. During the major period of the ‘Great Enclosures’ (1750–1850) hedges were typically established using a single species (hawthorn, *Crataegus monogyna*) and ran to some 321,869 km, at least doubling the entire length planted over the previous five centuries (Rackham 1986). However, ‘enclosure’, as a legal term, is not just the establishment of field boundaries; it also means that land can be farmed ‘in severalty’, i.e. independently of others, and requires the quashing of the rights of others over the land. That informal and formal legal and physical process had, of course, been going on for centuries prior to the Parliamentary Enclosure Acts and continued alongside them and into the 19th century (Hollowell 2000). As to present times, it is usual to plant multi-species hedges to maximise biodiversity benefits (Anonymous 2008a).

**Change in extent, status and condition of hedgerows and other field boundaries**

Since the end of World War II, the intensification of farming, land consolidation programmes, urban expansion, opencast mining and commercial and infrastructure development (Baudry & Burel 1984; Dowdeswell 1987; Fry 1991) has led to a substantial loss of hedgerows and other field boundaries. Land consolidation is the process whereby disparate landholdings, spread across a landscape, are collectively reorganised into contiguous ownership blocks; intensification, including hedgerow and other field boundary removal, typically follows. For example, a land consolidation programme in Brittany in 1982 subsequently led to a 50% reduction in hedgerows (with negative consequences for linnets *Acanthis cannabina* living in nearby heathland; Eybert et al. 1995).

Perhaps the most well-documented loss of managed hedgerows comes from the UK. Pollard et al. (1974) and Hooper (1981), citing (Locke 1962) suggested the stock of hedges in England, Scotland and Wales (Great Britain, GB) to be about 804,672 km ‘at the end of the 1950s’. Dowdeswell (1987) indicated 230,000 km of hedge had been lost between 1946 and 1974, presumably using the estimated
average annual loss of 8,047 km given in Pollard et al. (1974) and Hooper (1981). However, these early estimates cannot compare in accuracy with those from the UK Countryside Survey, which was initially based on a random stratified land-use survey of GB. Over time, the survey increased its scope to cover the whole of the UK by including Northern Ireland, and the number of sample squares was also increased due to this and changing policy requirements. So whilst 256 squares were surveyed in GB in 1978, 384 squares were examined in 1984, 508 in 1990, 569 in 1998 and 591 in 2007. Hence estimates have been based on increasing sample sizes over time (Table 1.1). The results indicated that between 1978 and 1984 some 28,200 km of managed hedge was lost from GB compared with a gain of just 3,600 km (e.g. from new plantings), but over the period 1984–1990 the net loss was considered to be 121,000 km (174,300 losses vs. 53,300 gains; Barr et al. 1991). Not all losses were removals, as a high proportion of ‘lost’ hedges changed from the definition used for managed hedgerows to a different category such as a line of trees (of the 174,300 losses, 111,500 were a result of changing boundary type). The definitions of different boundary types used by the Countryside Survey have been modified over time, so data given in the most recent report (Carey et al. 2009b) is not completely compatible with earlier reports (e.g. Barr and Gillespie (2000) but the trends remain the same. The 1998 survey period suggested that there was a slight gain in the stock of hedges over the period 1990–1998; however, data from the last survey in 2007 showed a further decline in the extent of managed hedges (Table 1.1). If we use the rough estimate of the length of hedges from Pollard et al. (1974) and the stock of hedges from the latest Countryside Survey (Carey et al. 2009b) (and accept that the result is going to be easily criticised on the basis of different methodologies), then it appears that since the late 1950s GB has lost about 41% of its net stock of managed hedgerows (i.e. after the balance between losses of existing and the planting of new hedges is taken into account).

Whilst the emphasis of this book is on hedgerows, it is clear that field boundary stocks are dynamic, with losses and gains recorded for different field boundary types. In GB, between 1984 and 2007, the total length of woody linear features (which includes hedges) has declined relatively modestly from 710,000 to 700,000 km (−1.4%), and walls have declined from 198,000 to 174,000 km (−12%). On the other hand, the lengths of banks/grass strips have increased from 56,000 to 64,000 km (+14%), and fences increased from 571,000 to 664,000 km (+16%). The length of lines of trees has rocketed from 32,000 to 114,000 km (+256%) – partially reflecting the cessation of management of many hedges such that they are now considered to be lines of trees/relict hedges (Carey et al. 2009b).

The herbaceous vegetation of hedgerows has also undergone drastic change over time, with significant decreases in species richness between 1978 and 1998 – though no subsequent change was detected by the Countryside Survey in 2007. Species composition also changed, with the proportion of more competitive species and those characteristic of shaded, fertile or less acidic soil conditions increasing between 1978 and 2007 (Carey et al. 2009b). The increase in shade tolerance of the herbaceous community was considered to reflect the increase in unmanaged
Introduction to hedgerows and field margins

hedgerows. These general trends were also reflected in a 70-year study based on 357 hedgerow sites in southern England by Staley et al. (2013). At the level of the individual hedgerow, species richness increased (for both woody and herbaceous flora), although different sites became more similar (homogeneous) over time. At the regional level, the herbaceous community of hedgerows declined over time, whilst the shrubby species composition increased (see further Chapter 3).

Landscape issues

How long is a hedgerow?

This somewhat prosaic question is actually quite important, especially when trying to count hedgerows and measure changes in extent. Is a hedgerow, for example:

- the total uninterrupted length of a hedge irrespective of the number of adjoining hedges or other field boundaries,
- the distance between two abutting hedges on any particular side,
- or the extent of a particular land use along its length on any particular side?

Each of these will give different answers, and each will be equally valid depending on the context and questions being asked of the dataset (Baudry et al. 2000). For connectivity, the density of hedges in an area, the frequency and extent of gaps and the number of connections between hedges (and with other landscape elements) may be the most important parameters.

Fragmentation and associated landscape ecology terms

When the concept of island biogeography theory (MacArthur & Wilson 1967) was first applied to terrestrial landscapes, the latter were conceptualised as binary systems only containing habitat (patches) and non-habitat (matrix). In such a system the matrix was considered devoid of resources and unable to support dispersers

<table>
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<tr>
<th>Survey period</th>
<th>Length of managed hedges</th>
<th>Number of 1 x 1 km sample squares compared</th>
<th>Cumulative loss (km)</th>
<th>Cumulative loss (%)</th>
</tr>
</thead>
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<td>Late 1950s</td>
<td>@804,672</td>
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<tr>
<td>1978</td>
<td>648,600</td>
<td>256</td>
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<td>3.9</td>
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<td>1978–1984</td>
<td>624,000</td>
<td>256</td>
<td>24,600</td>
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<td>1984–1990</td>
<td>506,000</td>
<td>340</td>
<td>142,600</td>
<td>21.7</td>
</tr>
<tr>
<td>1990–1998</td>
<td>508,000</td>
<td>508</td>
<td>140,600</td>
<td>21.7</td>
</tr>
<tr>
<td>1998–2007</td>
<td>477,000</td>
<td>569</td>
<td>171,600</td>
<td>26.6</td>
</tr>
</tbody>
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from habitat patches. Further elaboration of the concept recognised the existence of a wider suite of landscape elements (rather than just habitat), introduced corridors and stepping-stones that promote movement, barriers that retard it and later a realisation that the matrix itself contained resources and could be better visualised as being variably permeable (spatially and temporally; see Dover and Settele (2009) for an overview and source references). The intensification of agriculture since the 1940s caused extensive habitat loss and ‘fragmentation’ of semi-natural habitat, leading to the need to understand how fragmentation affected species at the local and landscape scale and how to mitigate its effects.

What we loosely term ‘fragmentation’ has distinct stages; starting from a notional homogenous habitat we can recognise four distinct states:

- ‘Intact’ habitat can be considered as a relatively homogeneous block containing 10 % or less non-habitat;
- ‘Variegated’ habitat would have 11–40% non-habitat;
- ‘Fragmented’ habitat would have 41–90% non-habitat;
- ‘Relictual’ habitat would have 91% or greater non-habitat (Hobbs & Wilson 1998).

Nor is fragmentation a single, one-off, process of habitat destruction as, in addition, it includes:

- ‘Shrinkage’ the subsequent reduction in size of remaining fragments;
- ‘Attrition’ where fragments are subsequently lost.

The initial process of habitat destruction that leads to fragmentation, coupled with subsequent shrinkage and attrition all combine to increase the ‘isolation’ of habitat patches (i.e. the gap between remaining habitat increases). This has potentially devastating impacts on populations (Hobbs & Wilson 1998). If fragments get too small they may be unable to support a viable population (Brito & Grelle 2006) and be too far away to be reinforced from other patches or to be recolonised (Forman & Godron 1986; Fahrig & Merriam 1994). The concept of the ‘metapopulation’ (Levins 1970; Hanski 1999) thus becomes relevant. Metapopulations are landscape-scale structures which are composed of a number of separate habitat patches which interact such that even a small number of movements of reproductively viable individuals dispersing between them can maintain genetic integrity, restock vacant habitat that had gone extinct through catastrophic stochastic events such as fire and buttress flagging populations. The interaction between populations of the different patches thus increases viability (persistence) of the species in the landscape in the long-term (compared with the fate of a single habitat patch). Not all habitat patches within a given metapopulation may be equally viable, for example, because they are too small to maintain a population long term or because the habitat quality is low. Such variability is recognised in terms of source-sink dynamics, whereby some patches are net exporters of individuals (dispersers) whilst others are net importers.
of individuals (i.e. sinks). Whilst the presence of a sink habitat may seem fairly use-
less, its presence may ultimately contribute to metapopulation stability even if it
only maintains a population for a few generations or acts as a stepping-stone to
other more viable patches. If habitat patches in a population are destroyed, this may
lead to increased isolation of the remaining patches and threaten metapopulation
stability or viability. Maintaining the ability of species to move throughout the
landscape (and hence cross the ‘matrix’) is thus paramount.

Hedgerows can potentially act as wildlife corridors, helping to reduce the
impact of habitat fragmentation by joining remnant patches together (Diamond
1975). In the context of climate change adaptation, Lawton et al. (2010) considered
that hedgerow networks (Forman & Baudry 1984) may help species to track newly
available climate space and leave former habitat that had become unsuitable. Unfortu-
nately, it is clear that hedgerow networks have also become fragmented due to
field amalgamation, resulting in large gaps and a reduction in connections between
hedgerows (Barr & Gillespie 2000). Corridors can have negative as well as positive
benefits, potentially exposing dispersers to predation and promoting disease trans-
mission etc. (Simberloff et al. 1992; Laine 2004). Whether hedgerows are essential
or just useful for dispersers will depend on the individual species/species group and
its degree of specialisation (Dover & Settele 2009). Individual species’ responses to a
hedgerow as corridor will also depend on how features such as gaps are perceived,
i.e. the difference between ‘functional connectivity’ and ‘physical connectivity’. In
the former, gaps may exist, but if the species is happy to move across the gaps then
it is functionally connected even though the hedgerow is not continuous (With
1997). This also implies that if something is physically connected, it may not be
functionally connected if some attribute important to the target species is missing.
Contrariwise, stepping-stones (Forman 1995), small patches of habitat, which could
include small discontinuous runs of hedgerow or even individual hedgerow trees
(see Chapter 10), may also aid dispersal even though they are physically discontinu-
os. It is also worth remembering that a corridor for one species may be a barrier
for another (Dover & Settele 2009).

There has been a kind of implicit assumption that a physical area termed ‘habitat’
contains all the resources needed for a given organism, but this is patently absurd, as
anyone who considers the process of migration must acknowledge. Birds, for ex-
ample, may migrate huge distances because of temporal changes in food resources or
the need to move to somewhere that contains some other vital resource such as
nest sites (Mönkkönen et al. 1992). Thus, it is clear that species move between areas
of resources, and it is more useful to think in terms of a resource patch than a strictly
delimited area that contains all the resources needed by a species. Resources may
indeed overlap in distribution (i.e. all present in much the same physical space),
but they may also be spatially separated. This resource-based approach is one that
has been pioneered by Roger Dennis and co-workers (Dennis et al. 2003; Dennis
et al. 2006; Van Dyck 2010; Dennis 2015). So at a more local level, individuals of a
species may move between food, overwintering and reproductive sites either in dis-
crete time periods, e.g. between summer versus winter, or during the same period,
e.g. moving between areas high in food and areas high in reproductive sites (i.e. for non-substitutable, ‘complimentary’ resources; Dunning et al. 1992). Individuals may also move between areas containing the same resource if its presence in one area is insufficient (supplementary resources; Dunning et al. 1992). The two terms are typically used as ‘complementation’ and ‘supplementation’ respectively (see, for example, Ouin et al. 2004). Essentially, corridors and stepping-stones can be considered as gateways to patch-based resources.

**Ecosystem services**

Ecosystem services, the benefits that biodiversity delivers directly or indirectly to humans, are typically divided into four different categories:

- **Supporting services** – those that are fundamental to life and global functioning such as photosynthesis, soil formation, water and nutrient cycles
- **Regulating services** – processes involved in providing an environment that supports human life, e.g. climate regulation, pollution control, pollination
- **Provisioning services** – products derived from ecosystems
- **Cultural services** – economic and cultural/aesthetic/spiritual values of biodiversity

(EASAC 2009)

The importance of specific ecosystem services, in relation to hedgerows and field margins, depends very much on personal viewpoint/context, with different ‘stakeholders’ viewing the values of hedgerows in different ways and depending on their personal capacity. Oreszczyn and Lane (1999) illustrated this nicely by describing, from interviews, how an agricultural adviser might value the ecological aspects greatest in his/her professional capacity but, as an individual, value the landscape aesthetics equally. The ecosystem services delivered may also be location specific, for example, with cultural value higher where a hedge marks a Saxon parish boundary or for water quality delivery where hedgerows are located near streams or ditches (Cole et al. 2009).

From a very one-sided production viewpoint, hedgerows take up growing space, may harbour crop pests, shade crops, ‘steal’ water from adjacent crops, remove nutrients and cause compaction on ground along headlands due to vehicle movements during farming operations, all of which conspire to reduce yield; they also cost money to maintain (Teather 1970; McCollin 2000). To balance these perceived disbenefits, we know that field boundaries also reduce soil erosion, are a source for pest predators and parasites, provide pollination services and provide a host of other values relevant to crop production, landowners and society as a whole (Table 1.2). Indeed, in some places hedgerows have had to be re-introduced and other structures such as beetle banks developed (Chapter 7) to reintroduce values lost through field amalgamation. I recall visiting a farm in Cambridgeshire which had been purchased by a landowner with an aim of increasing farming efficiency. He also
<table>
<thead>
<tr>
<th>Type of service</th>
<th>Specific service</th>
<th>Function/examples of service</th>
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<tbody>
<tr>
<td><strong>Supporting services</strong></td>
<td><strong>Soil formation</strong></td>
<td>Soil formation</td>
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<tr>
<td>Photosynthesis</td>
<td>Production of oxygen</td>
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<td>Primary production</td>
<td>Production of chemical energy in the form of organic matter</td>
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<td>Nutrient cycling</td>
<td>Recycling of materials such as carbon, phosphorous, nitrogen</td>
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<tr>
<td>Water cycling</td>
<td>Recycling of water</td>
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<tr>
<td><strong>Regulating services</strong></td>
<td>Air quality regulation</td>
<td>Air pollution control, e.g. removal of particulates and nitrogen dioxide; prevention of agrochemical drift</td>
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<tr>
<td>Climate regulation</td>
<td>Sequestration of carbon dioxide, source of renewable energy, temperature and humidity control, wind shelter for stock and crops; energy efficiency (shading, wind shelter); moderation of urban heat island effect as part of green infrastructure</td>
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<tr>
<td>Water regulation</td>
<td>Run-off and flood control (ditches, banks, infiltration pathways, rainfall capture reducing peak flows, transpiration, and promoting evaporation); irrigation channelling</td>
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<tr>
<td>Erosion regulation</td>
<td>Soil retention (soil stabilisation by roots), windbreaks to reduce erosion (capture of blown/suspended soil particles)</td>
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<td>Water quality/purification</td>
<td>Removal of sediments and pollutants, prevention of agrochemical drift into watercourses</td>
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<tr>
<td>Pest control</td>
<td>Source of predators and parasites of crop and livestock pests</td>
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<tr>
<td>Pollination</td>
<td>Nectar, pollen and nesting resources for pollinators</td>
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<tr>
<td>Natural hazard reduction</td>
<td>Prevention of landslides and costal erosion; snow breaks for railways</td>
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<tr>
<td><strong>Boundaries and barriers</strong></td>
<td>Delimitation of ownership boundaries and containment of stock; prevention of agrochemical drift; defence in warfare</td>
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<tr>
<td>Sense of well-being</td>
<td>Impact of natural environment on human well-being, mental and physical health</td>
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</table>
had an interest in field sports. Field boundaries were removed, farm staff sacked and tied houses sold off; ploughing, planting, fertilising, spraying, and harvesting were all contracted in by the estate agents left in charge, and the only directly employed member of staff left on the farm was the gamekeeper (who had a radio to call the police with when illegal hare coursers intruded from London). The owner was confounded when he discovered there were few gamebirds left to shoot; hedges then had to be replanted.
Cole et al. (2009) screened the ecosystem services identified by the Millennium Ecosystem Assessment (Reid et al. 2005) and considered that 22 were relevant to English farmland conditions. Of these, they considered that 19 ecosystem services were delivered by hedgerows as part of an evaluation of UK agri-environment schemes, though in their analysis they excluded/aggregated some due to the context of their work. Some of these latter have been restored in Table 1.2, some added they did not consider relevant, as well as additional services not specifically identified. Analyses of ecosystem services often do not include biodiversity, which, of course, is a value in its own right (Cole et al. 2009). Many of the ‘regulating’ ecosystem services are mentioned in the following chapters, especially in relation to the suppression of agrochemical drift (e.g. Chapters 5 and 6), pest control (Chapter 7) and pollination (Chapter 9).

Conclusion

Hedgerow networks have become fragmented over time due to the pressure on agriculture to deliver increased levels of efficiency. Nevertheless, hedgerows are, in some places, the only semi-natural habitat left in some intensive farming landscapes both providing habitat for wildlife and delivering essential ecosystem services. The loss of farmland biodiversity has led to attempts to counter this by the development and incentivisation of agri-environment measures and also through statute to reduce further loss. Nevertheless, hedgerows and field margins and the biodiversity they harbour face considerable challenges now and, with issues such as climate change, in the future, as will be evident from a perusal of the following chapters.

References


### APPENDIX 1.1

Studies aimed at improving the biodiversity value of arable field margins (earlier studies reviewed by Haaland et al. 2011)

<table>
<thead>
<tr>
<th>Main crop</th>
<th>Country</th>
<th>No. treatments</th>
<th>Duration</th>
<th>Margin management imposed</th>
<th>Organisms studied</th>
<th>Outcomes</th>
<th>Source</th>
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<tbody>
<tr>
<td>Field bean and Strawberry</td>
<td>Sweden</td>
<td>2</td>
<td>2011: Field bean 2013: Strawberry</td>
<td>1. Control: Crop only (strawberry or field bean) 2. Sown Wildflower strip + crop (strawberry or field bean)</td>
<td>Plants: Pollination success of strawberry <em>Fragaria vesca</em> and field bean <em>Vicia faba</em>. Pollinators of strawberry; flies, solitary bees + other Hymenoptera (not bumblebees or honeybees) and field bean: bumblebees + honeybees monitored by direct observation (experimental plants were in pots) but not analysed.</td>
<td>Landscape factors interacted with pollination success. Adjacent to the wildflower strips, pollination success was depressed in heterogeneous landscapes but enhanced in the distant crop edges. In more homogeneous landscapes pollination was enhanced near the wildflower strips and depressed at the distant crop edges.</td>
<td>Herbertsson et al. (2018)</td>
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<td><em>Strawberry and field bean studies were separate. Monitoring adjacent to sown wildflower strip, and at the edge of the crop &gt; 160 m distant; control sites monitoring at one field edge. Seed mixtures in wildflower strips varied.</em></td>
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<td>Arable (no specifics given)</td>
<td>Germany</td>
<td>2</td>
<td>2012–2013</td>
<td>1. Hedgerow + single wildflower strip in a 500-m-diameter area. 2. Hedgerow with several wildflower strips within 500-m diameter</td>
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**Hymenoptera:**  
**predatory solitary wasps**  
Foraging/provisioning of nests of two species studied in relation to proximity to wildflower strips using trap nests.

Species-specific and landscape composition differences were detected. *Anistrocerus nigricornis* (a hunter of crop pests: caterpillars) built more brood cells the closer the trap nests were to wildflower strips; but the number of brood cells also increased with a greater proportion of grassland in the study area. Fewer prey items were used to provision a cell in heterogeneous landscapes. *Trypoxylon figulus* (a hunter of pest predators: spiders) built fewer nests in proximity to wildflower strips. In sites with single wildflower strips, more cells were built in areas with higher levels of grassland; in sites where several wildflower strips were present, the lack of grassland was compensated for. For *T. figulus* wildflower strips were considered an adult rather than larval resource.

Hoffmann et al. (2018)
<table>
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<tr>
<th>Main crop</th>
<th>Country</th>
<th>No. treatments</th>
<th>Duration</th>
<th>Margin management imposed</th>
<th>Organisms studied</th>
<th>Outcomes</th>
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<td>Rape and winter</td>
<td>Belgium</td>
<td>5</td>
<td>2013–2015</td>
<td>1. Control: Grass (G) only sown.</td>
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<td>Pollinator richness was lowest in the control. Pollinator species richness and evenness were not positively affected by increasing levels of plant functional diversity in wildflower sowings and resulted in less flower visiting rather than more.</td>
<td>Uyttenbroeck et al. (2017)</td>
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<td>wheat</td>
<td>4. G + High FD WF</td>
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<td>5. G + Very high FD WF</td>
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<td>Functional diversity treatments created from a mixture of 7 plant species/treatment assessed on flower colour, flower type, UV reflection, presence of a UV pattern, flowering period, plant height from a pool of 20 species. A mix of 3 grasses also sown with the wildflowers.</td>
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<td>Invertebrates:</td>
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<td>Insects searching for floral rewards monitored via transects; pollinator networks monitored in permanent quadrats.</td>
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<td>Invertebrates:</td>
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<td>aphid predators (Chrysopidae, Coccinellidae; Syrphidae)</td>
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<td>High functional diversity (FD) of wildflower sowings did not correlate with increased abundance or richness of aphid predators. Coccinellidae were more strongly associated with low or intermediate levels of FD. Suggested that individual species, not present in all treatments, caused the response.</td>
<td>Hatt et al. (2017)</td>
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<td>Invertebrates sampled using pan traps.</td>
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<td>Winter wheat</td>
<td>Switzerland</td>
<td>2</td>
<td>2014</td>
<td>Invertebrates: cereal leaf beetles (Oulema sp.)</td>
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<td>Beetle egg and larval abundance (and crop damage) assessed at 5 m and 10 m from wildflower strips or equivalent width control (crop) margins by direct observation; adults sampled by sweep netting.</td>
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<td>Eggs and larvae of cereal leaf beetle and crop damage were reduced on wheat plants adjacent to wildflower sowings at 5-m but not 10-m distance. Wheat yields were higher at both 5- and 10-m distance where wildflower strips present.</td>
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<td>Tschumi et al. (2016a)</td>
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<td>Potato</td>
<td>Switzerland</td>
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<td>2013</td>
<td>Invertebrates: potato aphids (Chrysopidae, Coccinellidae, Syrphidae)</td>
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<td>Eggs were laid on potatoes adjacent to wildflower sowings and, aphid abundance on adjacent potatoes was reduced by 75%.</td>
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<td>Tschumi et al. (2016b)</td>
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<td>Crop</td>
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<td>Treatments</td>
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<td><strong>Invertebrates:</strong> cereal leaf beetles (Oulema sp.)</td>
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<td><strong>Invertebrates:</strong> potato aphids, Syrphidae, Chrysopidae and Coccinellidae</td>
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<td>Potato</td>
<td>Switzerland</td>
<td>2013</td>
<td>1. Control: winter wheat crop.</td>
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<td>2. Existing perennial wildflower (sown) strip</td>
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<td>Abundance of Syrphidae, Chrysopidae and Coccinellidae was greater in wildflower strips than in potato crop controls as was syrphid species richness (coccid and chrysopid richness not assessed). More chrysopid and syrphid eggs were laid on potatoes adjacent to wildflower sowings and, aphid abundance on adjacent potatoes was reduced by 75%.</td>
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<td>Tschumi et al. (2016a)</td>
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<td>Tschumi et al. (2016b)</td>
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<tr>
<th>Main crop</th>
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<th>No. treatments</th>
<th>Duration</th>
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<th>Organisms studied</th>
<th>Outcomes</th>
<th>Source</th>
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<tr>
<td>Winter wheat</td>
<td>Switzerland</td>
<td>2</td>
<td>2005</td>
<td>1. Winter wheat with a sown grass margin.</td>
<td>Invertebrates: Coleoptera: Carabidae</td>
<td>Abundance of carabids (activity density) was higher in the crops than in the grass or wildflower margins. Following rarefaction analysis, sown wildflower strips had higher species richness than in wheat fields and sown grass strips. Sown wildflower strips and grass margins had different carabid communities to wheat crops.</td>
<td>Anjum-Zubair et al. (2015)</td>
</tr>
<tr>
<td>Winter wheat</td>
<td>Switzerland</td>
<td>2</td>
<td>2012</td>
<td>1. Control: Winter wheat crop (nominal 3 m margin).</td>
<td>Invertebrates: cereal leaf beetles (Oulema sp.) + natural enemies: predatory Hemiptera, Coccinellidae, Neuroptera, Syrphidae, Carabidae.</td>
<td>Larvae of cereal leaf beetle, emerging adults, and crop damage were reduced on wheat plants adjacent to wildflower sowings. The reduction in crop damage was greatest in the zone furthest from the wildflower sowing. Adults of natural enemies were higher in wildflower strips than in the equivalent control wheat margins. Predatory Hemiptera were more abundant in the crop</td>
<td>Tschumi et al. (2015)</td>
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<td>2. Annual wildflower sowing (3-m margin)</td>
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<td>Country</td>
<td>No.</td>
<td>Year</td>
<td>Treatments</td>
<td>Margin management imposed</td>
<td>Organisms studied</td>
<td>Outcomes</td>
<td>Source</td>
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<tr>
<td>Switzerland</td>
<td>2</td>
<td>2005</td>
<td>1. Winter wheat with a sown grass margin. 2. Winter wheat with a sown wildflower margin.</td>
<td>Invertebrates: Coleoptera: Carabidae</td>
<td>Abundance of carabids (activity density) was higher in the crops than in the grass or wildflower margins. Following rarefaction analysis, sown wildflower strips and grass margins had different carabid communities to wheat crops.</td>
<td>Anjum-Zubair et al. (2015)</td>
<td></td>
</tr>
<tr>
<td>Switzerland</td>
<td>2</td>
<td>2012</td>
<td>1. Control: Winter wheat crop (nominal 3 m margin). 2. Annual wildflower sowing (3-m margin)</td>
<td>Invertebrates: cereal leaf beetles (Oulema sp.) + natural enemies: predatory Hemiptera, Coccinellidae, Neuroptera, Syrphidae, Carabidae.</td>
<td>Beetle egg and larval abundance (and crop damage) assessed in two zones (0.5–10.4 m and 10.5–20.4 m) from the wildflower strips. larvae of cereal leaf beetle, emerging adults, and crop damage were reduced on wheat plants adjacent to wildflower sowings. The reduction in crop damage was greatest in the zone furthest from the wildflower sowing. Adults of natural enemies were higher in wildflower strips than in the equivalent control wheat margins.</td>
<td>Tschumi et al. (2015)</td>
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<td>Finland</td>
<td>2</td>
<td>2007–2010</td>
<td>1. Control: Reed canary grass crop/ permanent field boundaries/adjacent cereal crop. 2. Wildflower sowings. At each replicate, wildflower sowings were established in 6 different orientations/plot sizes. Five sowings used the same seed mixture, one was a monoculture of Centaurea jacea. Control plots had the similar orientations as the wildflower plots.</td>
<td>Invertebrates: bumblebee abundance + species richness of bumblebees, butterflies and day-flying moths. Abundance and species richness assessed on transects. Evaluations made separately of three aspects: bumblebee abundance (pollination), combined species richness of the 3 invertebrate groups (diversity) and the richness of habitat specialist butterflies (biodiversity).</td>
<td>Bumblebee abundance and species richness of invertebrates was enhanced in wildflower strips for the first 3 years then declined in year 4. Specialist butterfly abundance increased more slowly than the other indicators and was influenced by the proportion of forest cover in the landscape. Flower abundance was particularly important for maintaining bumblebee abundance; and re-sowing was probably required to maintain local flower quality over time.</td>
<td>Korpela et al. (2013)</td>
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<td>Main crop</td>
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</table>
2. Scarified+wildflower sowing.  
3. Scarified+wildflower sowing + full-rate graminicide (once annually).  
4. Scarified+wildflower sowing + full-rate graminicide (once, in Year 1 only).  
5. Scarified+wildflower sowing + full-rate graminicide (twice, in Year 2 only).  
6. Scarified+wildflower sowing + half-rate graminicide (once annually).  
7. Scarified+wildflower sowing + half-rate graminicide (once, in year 1 only).  
8. Scarified+wildflower sowing + half-rate graminicide (twice, in Year 2 only).  
9. Full-rate graminicide (once, in year 1 only).  
10. Full-rate graminicide (once annually). Treatments carried out on the outer 4 m of existing (planted 2004/5 depending on site) grass buffer strips  | **Invertebrates: spiders**  
Spider abundance and richness sampled using suction.  | Wildflowers established well in the sown plots and did best where both herbicide and scarification were applied. Trends in spider abundance were linked to scarification, herbicide, and sown wildflowers.  | Blake et al. (2013) |

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<table>
<thead>
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<th>Country</th>
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<th>Outcomes</th>
<th>Source</th>
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<tr>
<td>Arable rotation (UK)</td>
<td>4</td>
<td>2008–2009</td>
<td>1. Control: grass strip. 2. Full-rate selective graminicide applied to grass strip. 3. Soil scarification applied to grass strip + wildflower sowing. 4. Selective half-rate herbicide + scarification + wildflower sowing. Treatments carried out on the outer 4 m of existing (planted 2004) grass buffer strips.</td>
<td>Invertebrates: butterflies Abundance and species richness assessed on transects. Wildflowers established well in the sown plots and did best where both herbicide and scarification were applied. Butterfly abundance, diversity and richness were best where both scarification and graminicide were used with wildflower sowings and were also influenced by the species richness of the wildflower community that established.</td>
</tr>
<tr>
<td>Arable rotation (Switzerland)</td>
<td>2</td>
<td>2008</td>
<td>1. Control: extensive grassland. 2. Sown wildflower strips (1–7 years old)</td>
<td>Invertebrates: butterflies Abundance and species richness assessed on transects in wildflower strips and extensive grasslands. The abundance of butterflies was greater in wildflower strips than in extensive grasslands. Species composition is distinct between the two types of site, with commoner species dominating the wildflower strips. However, one habitat specialist was found only in the wildflower strips – its hostplant was uncommon in the meadows but present in the strips. Several species bred in the strips. Wildflower strips, flower abundance and the presence of forest nearby positively influenced species richness.</td>
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<tr>
<th>Main crop</th>
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<th>No. treatments</th>
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</thead>
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<tr>
<td>Arable rotation</td>
<td>UK</td>
<td>5</td>
<td>1999–2002</td>
<td>1. Control: arable crop.</td>
<td>Invertebrates: butterflies, bumblebees, <em>Ananeae</em>, <em>Opiliones</em>, <em>Coleoptera</em>, <em>Heteroptera</em>, <em>Formicidae</em>, <em>Chilopoda</em>, <em>Diplopoda</em>, and <em>Isopoda</em>.</td>
<td>Natural regeneration resulted in a species-poor, agronomically unacceptable sward with a high species turnover, unlike tall grass sowings, which were stable but species poor and dense. Wildflower sowings, which included legumes and grass-regulating hemi-parasites (<em>Rhinanthus</em> sp.), produced the richest plant community, excluded pernicious weeds and supported a rich and abundant pollinator and herbivore community and had a greater abundance of predatory species.</td>
<td>Pywell et al. (2011)</td>
</tr>
</tbody>
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