

Managing ecosystem services in broadacre landscapes: what are the appropriate spatial scales?

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Abstract. Over the past 200 years agriculture has expanded throughout Australia. The culmination of clearing and cultivating land at the farm scale has resulted in highly modified landscapes and a perceived loss of ecosystem services from pest control and pollination. We examine the literature: (i) to identify the appropriate spatial scale for managing pests, natural enemies and pollinators; and (ii) for evidence that farm-scale changes (due to agricultural intensification) across a landscape have resulted in a tipping point favouring pests and hindering pollinators. Although there is limited information to draw firm conclusions, the evidence suggests that actions undertaken on individual farms have an impact both on their neighbours and regionally, and that the culmination of these actions can lead to changes in population dynamics of pests, natural enemies and pollinators. For major pest species, there is reasonable evidence that grain growers may benefit from improved management and higher yields by implementing area-wide pest management strategies on a landscape scale in collaboration with growers of other crops that also share these pests. As yet, for natural enemies and pollinators there is little direct evidence that similar area-wide initiatives will have a greater effect than management strategies aimed at the field and farm level. Managing pests, natural enemies and pollinators beyond the scale of the field or farm is technically and socially challenging and will require a well defined research agenda, as well as compromise, balance and trading among stakeholders. We highlight critical knowledge gaps and suggest approaches for designing and managing landscapes for ecosystem services.

Introduction

Since the colonisation of Australia by Europeans, agriculture has expanded from the first meagre plantings in 1788 to more than 25 million ha (ABS 2007). Although the land area devoted to cropping (6% of agricultural production) is only a small fraction of the continent, it is concentrated in a few regions with favourable rainfall (ABS 2007). In these landscapes the changes have been dramatic. These include clearing of native vegetation (Hobbs and Saunders 1994), expanding area in crops grown year-round that ostensibly provides a smorgasbord of host plants for major insect pests, for example *Helicoverpa* spp. (Lepidoptera: Noctuidae) (Passlow 1986), converting previously unfavourable habitat into a highly desirable habitat for locusts (Clark 1950), and extensive use of pesticides. The culmination of farms making these changes has resulted in: (i) areas of natural, 'semi-natural' and highly modified landscapes of various patch sizes and degrees of isolation (Hobbs and Saunders 1994); (ii) reduced habitat heterogeneity (Benton *et al.* 2003); (iii) areas favouring pests, hindering natural enemies and pollinators, and threatening biodiversity; and (iv) in some cases a perceived 'tipping point' (Grodzins 1957) – a very particular threshold that is a culmination of a build-up of numerous small actions that together result in a big change. Numerous small-scale actions of clearing of native vegetation,

reduction in crop diversity, and insecticide use may result in abrupt shifts in ecological responses (Turner and Gardner 1991) causing deterioration of ecosystem functions such as pest suppression, pollination, and species interactions to create a tipping point. For example, the result of the culmination of farms planting only two or three crop types, all of which are host plants to pests, may result in an abrupt sustained increase in pest populations.

As a corollary to the tipping point, there is greater recognition that insect pest and pollination problems need to be considered beyond the crop boundary. Crops grown in regions of extensive agricultural production may be more susceptible to pests than crops grown in isolated locations (Schellhorn and Lawrence 2008). This may be due to more habitats supporting pests, but also the absence of non-crop habitats, which are thought to play a crucial role in maintaining populations of natural enemies in agricultural landscapes (Bianchi *et al.* 2007). For example, even small and isolated fragments of remnant vegetation are thought to play an important role in maintaining the ecosystem services (*sensu* products of nature that yield human wellbeing; Banzhaf and Boyd 2005) of pest control and pollination. Remnant patches may provide perennial habitat for natural enemies (Bianchi *et al.* 2006; Renchen 2006), feral and native pollinators (Heard 2001; Blanche *et al.* 2006), species of conservation interest (Sand and

New 2002), and, in some instances, a source of pests with alleles susceptible to insecticides, which assists in delaying insecticide resistance (Summerville 2004).

The benefit of natural pest control is widely recognised. Natural enemies suppress populations of a wide range of insect herbivores thus most of the potential pest species are controlled and do not reach outbreak levels in field crops. The value of the ecosystem service of pest control is estimated at more than US\$400 billion per year worldwide (Costanza *et al.* 1997), but there is only limited quantitative information for the Australian situation. As for pollination, several crops in broadacre environments benefit from pollination services provided by animals and reliance on pollinators varies from high (clover 90%, faba beans 50%, field pea 50%) to low (canola 15%, lupin 10%; Cunningham *et al.* 2002).

Here we focus on the impact of farm-scale changes across Australian broadacre grain landscapes on pest control and pollination services. We review the literature for evidence that a tipping point has occurred, and for indications of the appropriate spatial scale for managing pests, natural enemies and pollinators. Further, we identify solutions to reversing current problems by highlighting technical challenges. We conclude with suggestions on who and what needs to compromise in order for change to occur.

Methods

Our approach is to ask a series of questions from the literature. First, is there evidence that a culmination of farm-scale changes across a region has resulted in an abrupt and sustained change in populations of pests and beneficial insects (i.e. natural enemies and pollinators)? The type of evidence that is sufficient to show support for this phenomenon would be a non-linear relationship between insect population change and the spatial scale of the landscape change. Weaker evidence would be distribution records that show that pests are currently more widely distributed than in the past. Second, to what extent can the presence and abundance of pests and beneficials be explained by processes happening at the field, farm, regional and national scale? This question highlights our present understanding of processes that drive insect dispersal at various spatial scales, which might give us clues about the appropriate spatial scale for management. The evidence required to show support includes: (i) dispersal data that differentiates between a migratory and appetitive behavioural process; and (ii) overwintering location data. Finally, is there evidence that managing organisms at a regional level (area-wide management, AWM) is more likely to achieve better outcomes (e.g. lower pest populations and more pollinators) than managing organisms at crop or farm level? This last question assesses whether a landscape-level approach to management can potentially result in a better outcome than a crop or farm-level approach. Evidence required to show support includes information on whether large-scale coordinated management for pests or beneficial insects results in lower pest levels, fewer sprays, more beneficial insects than individual crops or farm-level pest management.

To answer the questions above, the spatial scale of these land use changes matters. In this paper, we distinguish between the field, farm, and regional scales as spatial units where management

action is taken, even though these units are imprecise. In broadacre grain-growing landscapes the average field size is 400 ha, with the average farm size being 1570 ha (ABS 2007). Regions may be defined by water flow, such as catchments or by administrative units, e.g. Liverpool Plains or Namoi Valley, New South Wales; Darling Downs, Queensland; mid north, South Australia. The culmination of small-scale changes in regions may include individually or in combination, the type, percent, and spatial arrangement of native habitat loss, and the type, percent, similarity, spatial arrangement and management action undertaken within the crops that replace cleared vegetation.

For each group of organisms (pests, natural enemies and pollinators) we look for examples from the literature which provide evidence for or against our questions. The underlying process of population change includes changes to immigration, birth, death and emigration. Large-scale clearing may allow for increased immigration of pests across a region, and decreased immigration of natural enemies and pollinators. In most cases, evidence is scarce because most research has been focussed at the field level. Therefore, we provide examples from overseas or circumstantial and weak evidence, or indicate that information is unavailable. However, this evidence-based approach allows us to identify gaps and future research directions.

Pests

We address two case studies of grain insect pests, a host generalist (*Helicoverpa* spp.) and a host specialist [*Plutella xylostella* (L.), Lepidoptera: Yponomeutidae], where a landscape-level or AWM approach may be beneficial.

(1) Is there evidence that a culmination of farm-scale changes across a region has resulted in increased pest pressure, and is the pressure abrupt and sustained?

Agricultural regions tend to specialise on a few crops selecting those that are well suited for soil type, water availability, climate and demand in the market place. For the highly polyphagous *Helicoverpa punctigera* (Wallengren) and *H. armigera* (Hübner), this creates a smorgasbord of host plants (Zalucki *et al.* 1986, 1994). Larval stages can feed and survive on cotton, sorghum, sunflower, chickpeas, lucerne, lupins, soybeans, tobacco, maize, wheat, tomatoes, lettuce, sweet corn, capsicum and various bean crops (Zalucki *et al.* 1986), with any one of them available throughout the year (Passlow 1986). However, the seasonal dynamics of both species is also driven by weather and migration (Maelzer and Zalucki 1999). Other lines of circumstantial evidence that point to changes in landscape characteristics as a driver for increased *Helicoverpa* spp. pest status are: (i) positive correlations between pest pressure and increased area under cultivation (Maelzer and Zalucki 1999); and (ii) historical records suggesting a geographic distribution far less than present day, which most likely coincide with the expansion of agriculture; see distribution records in Common (1953), Kirkpatrick (1961) and Matthews (1999).

The diamondback moth (DBM), *P. xylostella*, is a host specialist on plants in the Brassicaceae and the major pest of Brassica crops (Talekar and Shelton 1993). Probably the biggest culmination of farm-scale changes across a region that has

influenced *P. xylostella* is the steady increase of Brassica cultivation in Australia over the years, particularly canola production in Western Australia (see Furlong *et al.* 2008). Canola is a DBM host and populations can reach high densities in the crop. The area under canola has grown enormously from around 100 000 from the mid 1980s to ~1 300 000 ha by 2005. When a canola crop supports only a single moth per m² (and densities are usually much higher), this implies an increase of 13 billion DBM in ~20 years. As the canola crop is rotated with wheat, moths may migrate out from such areas. The DBM problem in vegetable Brassica appears to have increased in southern States since about the mid 1990s (see Furlong *et al.* 2008), and coincides with a fairly steady increase of adult DBM numbers colonising Tasmania (Fig. 1). The exception to the increase is 2002–03, which may be due to several factors – one being wind.

(2) How much of pest abundance is due to processes happening at the field, farm, region and national scale?

Influxes of moths may occur into an area from a national scale, and from nearby fields. Both *Helicoverpa* species are migratory and can move many hundreds of kilometres among regions, and they can move extensively among fields within regions (Wardhaugh *et al.* 1980; Farrow and Daly 1987; Gregg *et al.* 1995; Rochester *et al.* 2002; Scott *et al.* 2005a, 2005b). Each year around mid-to-late September a synchronous peak of moths of both species occurs throughout eastern Australia (Fitt *et al.* 1989). The majority are *H. punctigera* carried by suitable wind systems from inland Australia into eastern cropping areas, but they rarely overwinter here (Gregg *et al.* 1995; Rochester and Zalucki 1998). However, the majority of *H. armigera* begin to emerge in the eastern cropping region from winter diapause (Fitt and Daly 1990), accounting for less than 5% in inland areas (Zalucki *et al.* 1994). In contrast to *H. punctigera*, *H. armigera* tends to increase in abundance during the cropping season, and those that

remain into autumn enter diapause and in the southern cooler areas over winter as pupae.

Other factors influencing local *Helicoverpa* abundance include weather, the degree of insecticide use, and the sequence and types of crops grown. Rainfall at the appropriate time and place drives the abundance of host plants within cropping areas (Zalucki *et al.* 1994). Maize, sorghum and lucerne have all been shown at some time to have a positive effect on *Helicoverpa* spp. abundance (Wardhaugh *et al.* 1980; Wilson 1983; Maelzer and Zalucki 1999). From a comprehensive series of partial life-table measurements, Titmarsh (1993) suggested that the relative contribution of plant hosts to local *H. armigera* numbers in a good season would be, in decreasing order: sunflower > chickpea > cotton > sorghum > mungbean > pigeonpea > maize > soyabean. Crop sequence varies with regions, indicating that landscape-level effects may be specific for regions. There is also uncertainty surrounding the relative contribution of wild host plants for *Helicoverpa* abundance, creating further problems for AWM planning.

For DBM, despite research conducted in a range of areas (see Furlong *et al.* 2008) long-term and landscape-level population dynamics has received surprisingly little attention. As a host specialist (e.g. Liu *et al.* 2005) one might expect abundance to be more tightly related to the availability of hosts, as well as to variation due to climate and migration (Zalucki and Furlong, in press). DBM can migrate large distances (Chapman *et al.* 2002), can move locally between properties (Schellhorn *et al.* 2004) and within a crop (Schellhorn *et al.* 2008), but can also be highly sedentary (Mo *et al.* 2003). This implies that DBM management should be considered at multiple spatial scales.

Host plant abundance and timing of planting are associated with DBM problems in vegetable-growing areas like south-east Queensland. Brassica vegetable production steadily increased in the Lockyer Valley with year-round production in the 1970s. Emerging DBM problems which ensued were ‘controlled’ with

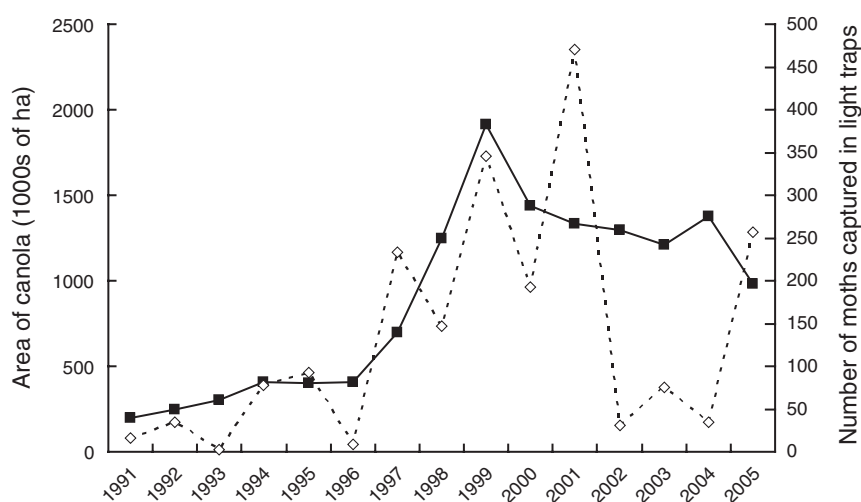


Fig. 1. Area of canola (solid line) grown in Australia and the number of diamondback moths (dashed line) caught in light traps in Tasmania, Australia, in spring. Crop area is based on Australian Bureau of Statistics data. Light trap data was provided by Lionel Hill, Tasmanian Department of Primary Industries and Water.

pesticides until resistance developed, causing major yield losses in the 1980s. This led to a rotation of pesticides, wider adoption of monitoring and spraying-type integrated pest management (IPM) with a summer production break. The latter has been credited as the circuit breaker, and the major driver of decline in DBM population density. However, Zalucki and Furlong (in press) point out that the break coincided with unfavourable climatic conditions for DBM population growth (Fig. 2). At this point, the relative contribution of landscape and climatic factors driving DBM population dynamics is unclear.

(3) Is there evidence that management of Helicoverpa or DBM at a regional level results in lower pest populations than management at individual crop or farm level?

AWM includes a range of tactics (e.g. release of pheromones to disrupt mating and mass trapping of moths, altered planting dates of crops, and large-scale release of natural enemies and sterile males) and tends to be pest specific and control method and timing dependent. The key to AWM is that the treatment is applied at a very large scale in a somewhat synchronous manner (Levins 1969) such that population increase due to immigration into the treated area is no longer significant and/or the population level has been reduced to such a low level it will fail to reach damaging densities during the growing season, even if growth rates are maximal (Knippling and Stadelbacher 1983). More recently, AWM has come to mean cooperative management programs implemented on a regional basis (Brewster *et al.* 1999). The essential feature is that by changing aspects of the landscape (e.g. reducing the number of host fields or the number of host fields with the pest) key processes leading to pest population increase will be disrupted.

Apart from Rochester *et al.* (2002), the strongest recent evidence that landscape-scale changes in crop selection can

affect abundance of pests has been the widespread growth of transgenic (*Bt*) cotton (Ingard[®] and more recently the two-gene Bollgard[®] cotton). In Australia, ~500 000 ha of cotton are grown in a season with close to 90% *Bt* cotton. This effectively makes genetically modified cotton a population sink, and there has been a decline in insecticide sprays for *Helicoverpa* (Knox *et al.* 2006) and an increase in natural enemy abundance (Mansfield *et al.* 2006). This period also coincides with a decline in climate suitability for *Helicoverpa* spp. (Zalucki and Furlong 2005), again making it difficult to isolate landscape-level processes.

There have been few attempts by the vegetable industry to manage DBM at a scale beyond the crop or farm. More recently, the grains industry has also shown interest in this topic. The initial exploration of the potential of such a strategy may best be accomplished using modelling because monitoring over large spatial scales is costly (see below). *Helicoverpa* spp. and DBM make for a nice comparison and contrast. The pests' ability to migrate means that factors external to the region will play a role, but there is clearly merit in managing pests at a regional scale. Although there is evidence that point to increases in pest problems with an increase in the area of host plants, evidence for a tipping point is not apparent.

Natural enemies

(1) Is there evidence that a culmination of farm-scale changes across a region has resulted in decreased natural enemy populations, and is the decrease abrupt and sustained?

There is little empirical evidence showing a decline in natural enemy populations. This is not surprising because such shifts are likely to happen over considerable time scales (e.g. decades) and require costly monitoring programs. However, the best documented example is the abrupt decline of ladybeetles in the

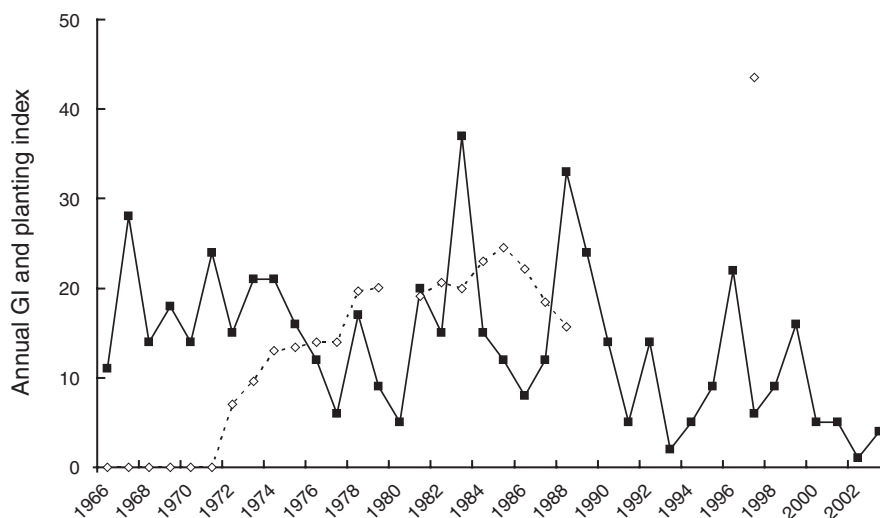


Fig. 2. The annual value of GI (solid line), a population growth index from CLIMEX for diamondback moth (DBM) based on climate data from Gatton, Queensland, Australia, for the period 1966–2003. The series crudely represents the climate suitability and abundance of DBM. The dashed line is the area of Brassica crops grown in the Gatton and Laidley Shires for years when data could be found. Note the severe outbreak in the early and late 1980s when the three-valley management strategy and production break was launched. Conditions appear to have been climatically unsuitable since then.

Czech Republic monitored over a period of 28 years (Bianchi *et al.* 2007). In this study, a spatially explicit simulation model was used to assess whether the strong decline of ladybeetles in the Czech Republic from 1990 to 2005 could be explained by changes in land use that occurred in the late 1980s. Simulations indicated that the replacement of 50% of the alfalfa acreage by rape had no detrimental effect on ladybeetle populations, but that the 50% reduction of fertiliser use and the replacement of 15% of the cereal acreage by rape resulted in lower aphid population densities at the landscape scale, and had a negative effect on energy sequestration, survival and reproduction of ladybeetles (Bianchi *et al.* 2007).

Circumstantial evidence of the role of hedgerows and forests for sustaining natural enemies comes from Germany where landscape mosaics composed of forests, arable crops and networks of hedgerows provide long-term and stable natural pest control. The forests and hedgerows host abundant specialist herbivore populations, which in turn support an abundant and diverse natural enemy complex that keep aphid densities in crops below economic thresholds (Ohnesorge and Schier 1989; references cited in Schulze and Gerstberger 1993). Thus, the limited (and circumstantial) evidence suggests that a tipping point most likely occurred in the Czech Republic example, but for other systems, there is a clear need for further study.

(2) How much of natural enemy abundance is due to processes happening at the field, farm, region and national scale?

Several studies have documented the occurrence of predators and parasitoids of agricultural pests in a range of crops (Johnson *et al.* 2000) and non-crop habitats in Australia (Rencken 2006; Stephens *et al.* 2006). Immature stages of insect predators have been found on a range of native plants indicating that these plants are locations of source populations (Table 1). On-farm crop refuges that are free from insecticides have been shown to host parasitoids that then move into the crop at distances of 120 m (Schellhorn *et al.* 2008). In addition, several of the key natural enemies have been shown to move between habitats at distances greater than a few hundred metres (Schellhorn and Silberbauer 2002; Silberbauer *et al.* 2004) or even between properties of

>1 km (Schellhorn *et al.* 2004). However, there is no information from Australia on natural enemy movement at a regional scale, and only the United Kingdom study by Chapman *et al.* (2006) has shown the seasonal migration of green lacewings (Neuroptera: Chrysopidae), a key predator of aphids, of distances up to 300 km. Although there is a considerable number of studies that suggest that natural enemies are influenced by landscape composition (Bianchi *et al.* 2006), there is limited information on natural enemy dispersal and migration, with the exception being ground-dwelling carabids (Coleoptera: Carabidae) (e.g. Dennis and Fry 1992).

Several studies have attempted to remedy the ephemeral agricultural landscapes by manipulating habitat within and adjacent to fields; several focussing on the effect of flower strips or field margins to enhance natural enemies in adjacent crops (e.g. Dennis and Fry 1992; Baggen and Gurr 1998; Tylianakis *et al.* 2004; Schellhorn *et al.* 2008). Yet, even a simple reduction in insecticide application frequency can result in big increases of pest mortality caused by natural enemies. Furlong *et al.* (2004a, 2004b) found much higher predation and parasitism rates in IPM than chemical control fields.

(3) Is there evidence that managing natural enemies at a regional level results in higher natural enemy pressure (or populations) than managing natural enemies at crop or farm level?

As multi-habitat use is common by natural enemies, one might expect that management to increase abundance and activity of natural enemies at a landscape scale will further complement management implemented at the field scale. However, there is little empirical evidence for this hypothesis. Although area-wide release programs with parasitoids (Vargas *et al.* 2004) and pathogens (Bell and Hayes 1994) have been conducted, the effectiveness of these programs are often difficult to assess because treatment and control areas are confounded by differences in temporal and spatial dimensions (Byers and Castle 2005). A simulation study focussing on the area-wide control of the boll weevil (*Anthonomus grandis* Boheman) with an exotic parasitoid, *Catolaccus grandis* (Burks), suggests that a slight improvement of pest control can be achieved when parasitoids are released at multiple locations or in a strip

Table 1. Immature stage of insect predators of agricultural pests found on native plants
Plants were sampled for natural enemies from December 2003 to April 2004 according to Stephens *et al.* (2006)

Plant species	Insect species
Chenopodiaceae	
<i>Atriplex semibaccata</i> (R.Br.)	<i>Nabis kinbergii</i> (Rueter) – damsel bug <i>Geocoris lubra</i> (Kirkaldy) – big-eyed bug <i>Coccinella transversalis</i> (F.) – transverse ladybeetle <i>Micromus tasmanica</i> (Walker) – brown lacewing
<i>Maireana brevifolia</i> (R.Br.)	<i>Nabis kinbergii</i> , <i>Micromus tasmanica</i>
<i>Rhagodia parabolica</i> (R.Br.)	<i>Mallada signata</i> (Schneider) – green lacewing, <i>Micromus tasmanica</i>
<i>Enchylaena tomentose</i> (R.Br.)	<i>Mallada signata</i> , <i>Nabis kinbergii</i> , <i>Micromus tasmanica</i>
Myrtaceae	
<i>Baekea behri</i> (Schldl.) F.Muell.	<i>Mallada signatus</i> , <i>Nabis kinbergii</i> , <i>Micromus tasmanica</i>
<i>Kunzea pomifera</i> (F.Muell.)	<i>Nabis kinbergii</i>

(e.g. by aerial release) than when released at a single location (Legaspi *et al.* 1998). In a theoretical study, Ives and Settle (1997) suggest that asynchronous planting of crops in a region will generally result in reduced pest densities in crops as opposed to synchronous planting. Asynchronous planting will in most cases lead to an early introduction of natural enemies (primarily if they are good dispersers) in crops at a time that they have a large impact on pest densities, which is likely to lead to effective pest control throughout the crop cycle.

These limited numbers of studies suggest that management to enhance natural enemies at the landscape scale can further complement management at the field scale. However, most of the evidence is based on theoretical studies. A solid underpinning of these model predictions by empirical studies is needed for a thorough evaluation of these predictions. This evaluation can then be used as a case study in the management of ecosystem functions at the landscape scale.

Pollinators

(1) Is there evidence that a culmination of farm-scale changes across a region has resulted in decreased pollinator abundance and pollination that is abrupt or sustained?

Crops that require insect pollination in Australia (e.g. some grain crops) rely heavily on the activities of feral colonies and commercially managed hives of *Apis mellifera* (Hymenoptera: Apidae), the introduced European honeybee. Native bees and other insects such as stingless bees, hoverflies, and nitidulid beetles do visit some crops but their role as pollination service providers is unknown (Cunningham *et al.* 2002). In many parts of Australia, 80–90% of native vegetation has been removed from agricultural landscapes and this has resulted in a loss of pollination services from birds and possibly insects (Paton 2000). Remnant vegetation and perennality is crucial for the survival of feral populations of pollinators, hence ‘free’ pollination services, because remnants provide suitable nesting habitat (e.g. hollows) and important food (Cunningham *et al.* 2002). The loss and greater isolation of native vegetation remnants and semi-natural areas on farms does reduce pollination services (Kremen *et al.* 2002, 2004; Blanche *et al.* 2006). In addition, our current agricultural practices do not value the free pollination services provided by insects as the wide-scale use of insecticides kill them. These factors combined with our reliance on a single species of pollinator increases our vulnerability to future disturbances, particularly with the threat of the mite, *Varroa destructor* (Acari: Parasitidae), that has decimated *A. mellifera* populations in many parts of the world (Cook *et al.* 2007). The effect of these factors may be concealed due to our ability to manage pollination by provision of quality hives at sufficient spatial distribution within crops. However, this can only be done to a certain point. For example, canola yields drop as the distance from the hives increases (Manning and Boland 2000). Conversely, the ability of the pollination industry to provide quality hives depends on access to suitable apiary sites in native vegetation for overwintering. Whether a tipping point has occurred in Australia is unclear; however, there are indications of reduced pollination service, at least at a local scale. In Europe, there is evidence of a national-scale decline in

pollinator diversity (density was not estimated) resulting in a subsequent decline in bee-dependent plants and increase in bee-independent plants (Biesmeijer *et al.* 2006). Several factors have been suggested as the cause for these declines, the most influential being agricultural intensification and habitat fragmentation (Carvell *et al.* 2007).

(2) How much of pollinator abundance is due to processes happening at the field, farm, region and national scale?

For pollinators, as with other functional groups, deterministic processes can occur at a range of spatial scales. For bumblebees the spatial scales at which individuals perceive their environment has been shown to be related to their body or colony size (Westphal *et al.* 2006). Bumblebees with medium-sized colonies were most affected by decreased landscape heterogeneity in agricultural landscapes (Westphal *et al.* 2006; Rundlöf *et al.* 2008). These colonies have medium-sized foraging ranges and forage resources within their range are not suitable for colony persistence. Species with larger ranges are able to forage from a much wider area and so are able to survive in more homogeneous landscapes. So for each pollinator species the influence of processes happening locally *v.* regionally will alter depending on the spatial scale with which they perceive their environment and the resources available.

Field and farm-level management, such as providing on-farm floral resources, less intensive farming practices, and improving the proximity to and quality of native remnant patches can provide benefits for pollinators. For example, the planting of field margins on farms with mixtures of nectar and pollen-producing plants is being implemented on a wide scale across the United Kingdom via the newly introduced environmental stewardship scheme. Studies have shown that flowering field margins can produce highly attractive forage resource for bumble bees in a relatively short period of time (Pywell *et al.* 2006; Carvell *et al.* 2007). However, the impact of such schemes on pollinator abundance and diversity has been shown to be heavily influenced by landscape-level factors (Heard *et al.* 2007). Less intensive farming systems that encourage within-field plant diversity and the prohibition of agrochemicals, such as organic farming (Roschewitz *et al.* 2005), may provide better habitat and foraging resources for pollinators. However, organic farms are only associated with higher species richness and abundance of bumblebees in homogeneous landscapes (Rundlöf *et al.* 2008).

Pollen analysis shows that pollinators do rely on native plant species growing in semi-natural habitats on farms (Williams and Kremen 2007). Furthermore, these semi-natural vegetation patches can improve and in some cases provide full pollination services to the adjacent crop (Kremen *et al.* 2002, 2004). The proximity of remnant patches is important because pollination services have been shown to decrease with increasing distance from patches. This has been demonstrated for honeybee visits to macadamia flowers in Australian orchards near rainforests (Blanche *et al.* 2006) as well as for honeybee and other pollinator visits to grapefruit flowers near remnants of subtropical Argentinean forest (Chacoff and Aizen 2006). These patterns are potentially less important in cereal crops that rely primarily on wind pollination. However, a study on

the primarily wind-pollinated *Brassica napus* (oilseed rape) showed that susceptibility to long-distance cross-pollination by wind depends on the level of bee activity within fields (Hayter and Cresswell 2006).

The quality (presence and density of flowers and nectar volume) of the remnant vegetation may also be important for pollination (Westphal *et al.* 2006). Native vegetation remnants within agricultural landscapes are often the sole remaining patches of perennial vegetation, and the quality is generally fairly poor due to the removal of larger trees, grazing by introduced herbivores and invasion by introduced weeds (Paton 2000). This may result in lack of flower availability during certain times of the year, causing periods of food limitation for pollinators (Paton 2000). Westphal *et al.* (2006) showed that in environments with poor resources (scattered vegetation remnants with low food-plant density) foraging trips were longer than foraging trips in environments with abundant resources. Furthermore, colonies in environments with poor resources also gained less weight. Perennial vegetation remnants that provide high-quality resources throughout time will result in greater pollinator abundance and activity.

(3) Is there evidence that managing pollinators at a regional level (e.g. AWM) results in higher pollinator activity (or populations) than managing pollinators at crop or farm level?

There is little evidence to suggest that managing pollinators at the regional level will result in greater pollinator activity. Studies which have attempted to disentangle the impact of environmental factors on arthropod communities across a range of spatial scales have concluded that local community composition is most affected by regional processes but highly confounded by landscape and local factors (Schweiger *et al.* 2005). Hendrickx *et al.* (2007) concluded that total landscape species richness of arthropods (including pollinators) was most strongly affected by proximity to native vegetation patches. For some pollinators there will be interacting effects of local farming practice and landscape context (Rundlöf *et al.* 2008).

Studies to date suggest that reducing farming intensity and increasing habitat connectivity at a regional scale will increase pollinator diversity and activity; however, this has yet to be demonstrated in the field. We conclude that local management options, if implemented on an individual farm, will advantage some pollinators and in turn provide some pollination services to that farm. However, the maximum benefits on a per farm basis may only be achieved through a more extensive and co-ordinated implementation of the same management option across an entire region.

Strategies to reverse the problems

The evidence presented above indicates that actions undertaken on individual farms have an impact both on their neighbours and at the regional scale, and that the culmination of these actions can lead to changes in population dynamics of pests, natural enemies and pollinators. Strategies aiming to reverse these changes need to accept that there will be technical and social challenges. Many

cross agricultural–environmental problems involve competition between several legitimate interests that require compromise, balance and trading across them to resolve. To design broadacre landscapes for multiple outcomes requires recognition of ‘what we are doing wrong’ (Brennan 2004) by the entire market chain from natural resource managers, primary producers (including those other than broadacre cropping), processors to consumers, and other land users (e.g. urban and tourism users). However, primary producers will probably be charged with implementation of landscape changes (with support from other sectors). Solving the technical and social aspects will require a well defined research agenda, as well as compromise, balance and trading among stakeholders. Here we concentrate on the technical challenges. Social challenges, which can be just as important as technical challenges, are beyond the scope of this study (see Kaine and Bewsell 2002; Brennan 2004; Kaine *et al.* 2005).

What are the technical challenges and solutions?

Pests

Dealing with a multivoltine, highly mobile species means that individual crop and farm based pest management is flawed, and that management implemented at larger spatial scales may be advantageous, but challenging. For *Helicoverpa* and DBM we have been essentially shepherding the problem around. The main question for such pests is: how large does a management unit have to be in order to be effective? *Helicoverpa* spp. and DBM need to be managed on an area-wide, season-long basis, which means that all growers in a region will need to cooperate if the pest population as a whole is to be reduced. For *Helicoverpa* spp. intervention, preferably with soft spray options such as specific virus (Gemstar[®]) or bacteria (DiPel[®]), in low value crops such as sorghum (depending on demand) during the season may be needed to reduce pest pressure on high input crops later in the season. Such a program appears to have been successfully implemented on the Darling Downs (Murray *et al.* 2005). For a host specialist such as DBM, host plant effects will be larger, but it may not be possible to simply reduce the area of cropping for production reasons. Managing pest populations in source crops such as canola or weedy *Brassica* has yet to be explored. Maximising natural enemy mortality by reducing broad-spectrum insecticide use would be a good start. If it works on a field scale there are likely benefits at larger spatial scales.

Natural enemies

In the Australian situation, relatively little is known about natural enemies compared with the pests they control. Several areas need further investigation before we understand the underlying processes governing natural enemy dynamics at large spatial scales let alone trying to make serious efforts in restoring landscape features as a means to enhance their abundance and effectiveness. A research agenda for the future should include: (i) identifying habitats that act as sources of natural enemies and pests; (ii) understanding their dispersal from these sources into crops; and (iii) acquiring quantitative understanding of tipping points and the spatial scales at which they operate.

Pollinators

There are many potential native pollinators in Australia that may be good candidates for supplying agricultural pollination services (Cunningham *et al.* 2002). Preliminary studies on native stingless bees suggest they could be useful low-maintenance pollinators (Heard 2001). Studies in the Americas have shown that native bees alone could provide sufficient pollination at over 90% of the 23 farms studied (Winfree *et al.* 2007). However, a lack of knowledge about their ecology and biology makes their management difficult at this point in time. To examine such trends in Australia, landscape data from coordinated regional sampling strategies over many years is required (Ghazoul 2005). These sampling strategies must focus on both crop and non-crop plants that are most vulnerable to pollinator losses, and highlight situations where crop failures are primarily due to a reduction in pollination services. Additional studies examining the level of pollinator redundancy for particular crops, variations in nesting and foraging resources across space, and pollinators' ability to move between patches across unfavourable matrix types (Kremen and Ricketts 2000) are all required to manage this ecosystem service.

Research methods needed to design agricultural landscapes for multiple benefits

There is a clear need for further understanding how changes in landscape composition impact on pest control and pollination at multiple spatial scales. As the manipulation of landscapes experimentally is generally not feasible, a modelling approach can be advantageous for addressing these kinds of questions. Mathematical models offer the possibility to unravel and study the processes (colonisation, reproduction, mortality and emigration) underlying ecosystem functioning and generate hypotheses that can be tested in empirical studies. A diversity of spatial models have been developed ranging from relatively simple analytical models (Corbett and Plant 1993) to complex simulation models (Potting *et al.* 2005). Models have been used to explore how the spatial arrangement of non-crop habitats affect pest control in crops (Bianchi and van der Werf 2003; Potting *et al.* 2005), how the timing of crop colonisation by pests affects the population dynamics of predators and ultimately pest suppression (Bianchi and van der Werf 2004) and how landscape diversity and heterogeneity affect spatial-temporal dynamics of spiders (Halley and Dempster 1996; Thorbek and Topping 2005).

When the purpose of the modelling is to generate realistic predictions of ecosystem functions it is essential that the key processes underlying the functions are incorporated. To reach this goal, an iterative modelling and experimental approach can be advantageous. Modelling may be used as a starting point with the objective to integrate existing knowledge resulting in a preliminary model which can be used to derive further hypotheses. Experimental studies can then be conducted to fill in knowledge gaps and test hypotheses. In the ideal case the end of this process would be a set of general rules of thumb that can be used to direct management strategies. Examples of such rules could be 'the pest control function is likely to be reasonably good in landscapes that contain more than 9% of non-crop habitat' (Bianchi and van der Werf 2003).

A challenge for the future is to design landscapes that are beneficial for a range of functions. As a first step, information (e.g. rules of thumb) is needed on how landscape composition affects each individual ecosystem service. Next, these rules of thumb should be integrated in a spatial framework to generate predictions for different landscape configurations and identify configurations that in the ideal case can lead to synergy between ecosystem services. The prospects for synergy between pest control and pollination seem good because some insects (e.g. hoverflies) provide both pest suppression and pollination services and important pollinator groups such as bees and many natural enemies of agricultural pest species depend on similar non-crop habitat. This suggests that management aiming at attaining a critical proportion of non-crop habitats in the landscape in a favourable configuration may be a promising way to enhance the ecosystem services of both pest control and pollination.

Conclusion

Designing landscapes for multiple outcomes is a technically and sociologically complex problem. In this paper we have provided an overview of evidence that suggests grain growers may benefit by implementing AWM strategies on a landscape scale in collaboration with growers of other crops that also host the major pest species. As yet there is little direct evidence that similar area-wide initiatives targeted for natural enemies and pollinators will have a greater effect than management strategies aimed at the field and farm level. We have suggested an iterative modelling and experimental approach to begin to address this question, and highlighted critical knowledge gaps that are hindering our ability to effectively manage landscapes for ecosystem services. Some of the strategies involved may also result in improved environmental outcomes such as enhanced amenity values and biodiversity conservation. Although not dealt with here, tools are available to assist with community engagement, system design, and policy development to drive implementation. Although farmers will probably be charged with the execution of landscape changes, the force to implement these changes may come from a range of stakeholders.

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