Pesticides and bees: Ecological-economic modelling of bee populations on farmland

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ABSTRACT

Production of insect-pollinated crops typically relies on both pesticide use and pollination, leading to a potential conflict between these two inputs. In this paper we combine ecological modelling with economic analysis to investigate the effects of pesticide use on wild and commercial bees, whilst allowing farmers to partly offset the negative effects of pesticides on bee populations by creating more on-farm bee habitat. Farmers have incentives to invest in creating wild bee habitat to increase pollination inputs due to the contribution this makes to yields. However, the optimal allocation of on-farm habitat strongly depends on the negative effects of pesticides, with a threshold-like behaviour at a critical level of the impairment. When this threshold is crossed, the population of wild bees becomes locally extinct and their availability to pollinate breaks down. We show that availability of commercial bees masks this decrease in pollination services which would otherwise incentivise farmers to conserve the wild pollinator population. Indeed, if commercial bees are available, optimum profit may be achieved by providing no habitat at all for wild bees, and allowing these wild pollinators to go extinct.

1. Introduction

Globally, around three-quarters of food crops are at least partly dependent on insect pollination (Klein et al., 2012), and this share has been rising over the past 50 years (Aizen and Harder, 2009). Ensuring sufficient pollination of these crops will be challenging in the future, due to adverse pressures on the supply of pollination services. Wild insect pollinator populations are threatened by habitat loss, declines in foraging resources (Carvell et al., 2007; Winfree et al., 2009) and agricultural intensification (Biesmeijer et al., 2006; Cameron et al., 2011), leading to population declines (Cameron et al., 2011; Goulson et al., 2015). For some crops, honeybees are used to supplement or substitute wild pollinators, along with other commercial pollinators such as factory-reared bumblebees (Velthuis and Van Doorn, 2006), although the majority of insect pollination for most crops is currently still delivered by wild pollinators (Breeze et al., 2011; Garibaldi et al., 2013).

However, whilst commercial pollinators can be substitutes for wild pollinators for many crops, (Brittain et al., 2013a,b), the use of commercial pollinators is not without risk. Honeybees have suffered losses in recent years due to the abandonment of hives (Colony Collapse Disorder), the impacts of the *Varroa* mite and associated diseases (Cox-Foster et al., 2007) and falling numbers of bee keepers in some countries (Potts et al., 2010). If losses of honeybees occur over a wide area, there can be an impact on the supply of these insects for pollination services, which can lead to cost increases to farmers; for example, prices for honeybee hire for use on almond farms doubled between 2006 and 2008 in the US (Pettis and Delaplane, 2010). Given the risks associated with reliance on commercial pollination sources, maintaining viable wild pollinator populations is likely to be crucial for sustaining the production of insect-pollinated crops into the future (Garibaldi et al., 2013; Winfree et al., 2007). Moreover, as we show in this paper, the availability of commercial bees can mask declines in wild pollinators past a local extinction threshold, threatening the supply of a wider set of valuable ecosystem services supplied by wild pollinators (Hanley et al., 2015).

One of the factors implicated in the decline of insect pollinators such as bees is the use of pesticides. There is growing evidence of negative effects of commonly used insecticides on population-determining traits such as foraging rates and navigation in bees, on the overall growth and performance of colonies, and on the pollination services that they provide (Mommaerts et al., 2010;

Henry et al., 2012; Gill et al., 2012; Whitehorn et al., 2012; Goulson, 2013, 2015; Rundlöf et al., 2015; Stanley et al., 2015). Awareness of this evidence has led to the temporary banning of the use on flowering crops of a widely used group of insecticides – neonicotinoids – within the European Union, but other insecticides are still widely used. Farmers of insect pollinated crops therefore face a dilemma, as one input (pesticides) is potentially dangerous to another (pollinators). One option, not investigated here, is to switch production to organic principles, and use zero pesticides. However, in the majority of global agricultural systems, abstaining from the use of all pesticides is not usually possible without substantial sacrifices in yields. Farmers must either attempt to reduce the impact of pesticides on wild pollinators, or increase the use of commercial pollinators which can be replenished year after year.

Wild pollinators require habitat either off-farm or within the farm area. Although pollinating insects can forage over large distances, in intensive agricultural landscapes there is a decay in visitation of flowers by pollinators with increasing distance from the nearest habitat patch (Ricketts et al., 2008; Osborne et al., 2008). To offset this, farmers can encourage wild bees to nest within foraging distance of crops by providing nesting habitat and alternative foraging resources on the farm for when the crop is not in flower (Carvell et al., 2007). The effect of such interventions has been found to be strongest in intensively farmed areas (Carvell et al., 2011) but depends also on the spatial location of bee-friendly habitat (Keitt, 2009; Brosi et al., 2008). Hence, local or field-scale management practices may offset the negative impacts of intensive monoculture agriculture on pollination services to some extent (Kennedy et al., 2013).

In this paper, we develop an ecological-economic model to investigate the relations between two agricultural inputs, pollination and pesticides, and two sources of pollination with different characteristics; commercial pollinators, which can be replaced at a cost, and wild pollinators, which rely on a population being sustained within the farm area. Dedicating some of the farm area to sustain wild pollinators (e.g. by cultivating wild flower strips) is assumed to be costly in terms of foregone profits from maintaining a larger cropping area (Breeze et al., 2014). The model is parameterised using farm management data for strawberries, a relatively well-studied crop on which both wild and commercial bees are used. The neonicotinoid pesticide thiacloprid is also commonly used in strawberry farming to protect the crop from destructive pests such as capsid bugs. Our modelling framework is, however, generalizable to other cropping systems where conflict occurs between pesticides, crop area and the survival of wild bee populations. Our model improves on previous modelling attempts which have looked at either habitat considerations (Keitt, 2009; Brosi et al., 2008) or pesticide impacts (Bryden et al., 2013) in isolation. In contrast, we combine these factors co-determining pollinator populations in a realistically-parameterised model which includes both economic and ecological behaviours.

2. Methods: the ecological economic model

The model has three main linked components: the dynamics of the wild bee population; a production function which links bee populations and pesticide use to output, and farmers' decisions over which inputs to employ, represented via a profit function. We consider a farm that produces a single crop; parameters are chosen to represent a typical soft-fruit production system (Nix, 2015; Ellis, 2014). The farm has an area A which is divided into a wild bee habitat conservation area, vA, and a cropping area $(1-v)A$, where v is the proportion assigned to the wild bee habitat (for modelling purpose we vary this between 0% and 70%). Honeybees and commercially reared bumblebees can both provide pollination services for fruit

production. For simplicity we consider all commercial (non-wild) pollinators to have the characteristics of commercially reared bumblebees in terms of nest size and pollinating efficiency, and generate results for both a scenario where all pollinators are affected by pesticides, and a scenario where wild bees are affected but commercial bees are not. These choices correspond to extreme situations; in reality it is possible that commercial pollinators are affected, but to a lesser extent than wild bees, since efforts can be made to minimise chemical exposure to commercial nests such as shutting the bees inside the boxes before spraying, or only spraying before the placement of nest boxes. Wild nests, on the other hand, may be exposed to multiple sprays of insecticides. Although both wild and commercial bumblebee nests are vulnerable to disease, wild nests are more likely to have infestations of parasites at the time spraying occurs (commercial bee boxes *should* arrive at the farms free from disease and therefore only pick up infections and parasites from that point onwards), putting wild bees at increased risk of any interactive effects between parasites and pesticides (Alaux et al., 2010).

For simplicity we are assuming that the farm is a closed system with regard to wild or commercial bees, so that bees are not migrating in from surrounding non-farmed habitat or leaving the farm. In reality bees do move between farms, which may buffer some of the more extreme effects predicted by our models (such as local extinction), and also means that bee populations supported by the actions of one farmer may benefit their neighbours. However, we do not capture the value of this external benefit in the model. We also assume no transfer of pesticides across the boundaries of the farm.

2.1. Wild bee population

The dynamics of the wild bee population is described in terms of $N(t)$ – a number of nests in a given year, t. This evolves according to Eq. (1):

$$N(t) = \min(R(N(t-1) - D(t-1)), K) \tag{1}$$

where $N(t-1)$ is the number of nests at the beginning of year t-1, $D(t-1)$ represents the number of nests that die during year t-1. $N(t-1) - D(t-1)$ represents the number of live nests at the end of year t-1 that will reproduce in the following year. R is the reproduction rate, i.e. the number of new nests that each reproducing nest produces in the following year. The carrying capacity, K, is calculated from the likely on-farm nesting densities of wild bumblebees, N_w, under the assumption that wild bees nest in the conservation area only, $K = N_w vA$. The simple, piecewise linear function, Eq. (1), captures the essential features of the nest dynamics – exponential growth for small numbers of nests, limited by a carrying capacity, K, for large numbers. We also considered alternative formulation of (1) with a logistic functional form; this produces very similar results, so they are not shown in this paper.

Not all bumblebee nests will produce queens in a given year, and the likelihood of reproduction will depend in part on nest size. Pesticides can indirectly impact the likelihood of a nest reproducing by impairing the performance of foragers or increasing worker mortality and thus decreasing a nests' ability to gather and process resources. These impacts can lead to increased colony failure, either through early colony death or by limiting the number of new queens produced (Gill et al., 2012; Whitehorn et al., 2012; Rundlöf et al., 2015). Bryden et al. (2013) suggested a model in which the probability of nest death was inversely proportional to the number of foragers adjusted for pesticide impairments. Here we use an