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Measuring the economic value of pollination services: Principles, evidence and knowledge gaps

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ABSTRACT

An increasing degree of attention is being given to the ecosystem services which insect pollinators supply, and the economic value of these services. Recent research suggests that a range of factors are contributing to a global decline in pollination services, which are often used as a "headline" ecosystem service in terms of communicating the concept of ecosystem services, and how this ties peoples' well-being to the condition of ecosystems and the biodiversity found therein. Our paper offers a conceptual framework for measuring the economic value of changes in insect pollinator populations, and then reviews what evidence exists on the empirical magnitude of these values (both market and non-market). This allows us to highlight where the largest gaps in knowledge are, where the greatest conceptual and empirical challenges remain, and where research is most needed.

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1. Pollination: a headline ecosystem service?

Animal pollination, usually via insects, birds or bats, influences the reproductive success of ~87% of flowering plants world-wide (Ollerton et al., 2011). Worldwide, ~1500 crops require insect pollination (Klein et al., 2007), and ~3 to 8% of global crop production (in tonnage) depends on insect pollination (Aizen et al., 2009). In temperate regions, most animal pollination is provided by honeybees (*Apis mellifera*), bumblebees (*Bombus* spp.), solitary bees, wasps and hoverflies, while in the tropics, butterflies, moths, birds and bats become important (Klein et al., 2007). Some crops, such as oilseed rape, are effectively pollinated by a broad range of insects, while others are specialized for pollination by particular insects; for example cocoa (*Theobroma cacao*) is primarily pollinated by midges (Klein et al., 2007). A number of bee species are actively managed, most notably the honeybee. Managed bumblebees are most commonly used in enclosed production systems (glasshouses and poly-tunnels), but other managed species are predominantly used for field and orchard crops (eg apples and almonds). Globally, evidence is emerging that wild bees and other insects are more important to crop pollination than managed bees (Garibaldi et al., 2013, 2011).

Since pollination is an ecosystem service which humans depend on through its link to world food production, it has become an often-cited example of how ecosystems services are economically valuable. The economic value flows from pollinators are both market and non-market valued. Market-valued benefits from pollinators consist of the contribution they make to the growing of a range of agricultural and horticultural crops (Gallai et al., 2009). Recent estimates suggest that crop pollination by insects underpins £430 million of crop production in the UK (Smith et al., 2011), with an equivalent figure of $361 bn world-wide (Lautenbach et al., 2012). However, there is considerable doubt over the precision, reliability, usefulness and interpretation of such figures. Non-market benefits derive from the utility which people derive from seeing pollinators or simply knowing they are being conserved and the indirect values derived from the aesthetic and cultural value of the wild flowers and garden plants which require pollination to sustain them. At any point in time, the present value of the future stream of market- and non-market valued benefits from pollinators, ie the value that can be derived in future, defines the value of this natural asset within a landscape.

The ecosystem service values derived from pollinators depend to a large extent on the condition and extent of the stock of pollinators, which is part of an area's natural capital. The value of

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pollinators as a natural capital asset depends on the stream of economic benefits which pollinators provide over time. However, in many areas, the ability of this natural capital asset to supply us with benefits has been diminished, due to pollinator population declines. Vanbergen et al. (2013) list the following main pressures on the supply of pollination services. These pressures, many of which are the result of economic activities, ultimately result in economic losses to the flow of ecosystem services from the stock of pollinators:

(1) Landscape change in agricultural landscapes: wild pollinators such as certain bumblebees may be disadvantaged from the loss of food sources due to decline in the area of wild flower meadows (Osgathorpe et al., 2011). More specialised pollinators tend to be more sensitive to the types of land use change inherent in land use intensification (Winfree et al., 2009). The increasing use of monocultures has been demonstrated to benefit wild pollinator abundances (e.g. Holzschuh et al., 2013) but can cause adverse community shifts (e.g. a reduction in long tongue bumblebees; Diekötter et al. (2010)) and may draw pollinators away from wild plants (Holzschuh et al., 2013). Increased synthetic fertiliser use and livestock stocking density can also cause significant long-term shifts in floral communities, reducing available forage resources for pollinating insects (Isbell et al., 2013; Hudewenz et al., 2012). On the other hand, farmer enrolment in agri-environment schemes which provide bee-friendly habitat will reduce the negative effects of agricultural landscape change (Scheper et al., 2013).
(2) Growing use of certain pesticides: there is evidence that insecticides such as neonicotinoids have significant non-lethal effects on both wild and managed bees, leading to reductions in foraging performance, decreased navigational abilities, reduced fecundity, and increased susceptibility to disease (e.g. Whitehorn et al., 2012; Di Prisco et al., 2013; Goulson, 2013). There is also growing evidence that contact with herbicides (Cousin et al., 2013), fungicides (Pettis et al., 2012) and even certain miticides (Berry et al., 2012) can have negative effects upon honeybee colony survival.
(3) The introduction of alien species: Invasive plants can have detrimental effects on native pollinators by displacing native flowers (e.g. Sugiura et al., 2013), although in some instances invasive plants species that are highly rewarding may benefit native pollinators: an example is the spread of Himalayan Balsam in Europe (Bartomeus et al., 2010). Invasive, non-native bees can displace native species either through direct competition or via spread of non-native diseases (Goulson, 2003; Arbetman et al., 2013).
(4) Pathogens and parasites. Pollinators suffer from a range of parasites (Vanbergen et al., 2013) and a range of bacterial, viral, protozoan and fungal diseases. The large scale anthropogenic movement of managed bees (primarily honeybee colonies and commercial bumblebee nests) has been linked with increased disease loads in the surrounding landscape (Meeus et al., 2011) and the spread of non-native parasites and pathogens against which they have little resistance (Graystock et al., 2013). The best known example is the mite *Varroa destructor*, accidentally introduced to Europe and the Americas from Asia.
(5) Climate change: climate change has been linked with changes in species range (Franzen and Ockinger, 2012) and growing mis-matches between insect emergence and floral bloom (Kudo and Ida, 2013). Which bees pollinate which crops in specific regions may also change. Honey bees are less vulnerable due to their managed status and the broad range of climates they can occupy, although their activity, and therefore service delivery, may alter (Rader et al., 2013). Climate change may also facilitate the growing of new insect pollinated crops in some regions e.g. the expansion of fruits northwards, but is also likely to result in the abandonment of some crops.

In this paper, we provide an overview of why the economic valuation of pollination services is useful to policy-makers and other stakeholders. This is followed by a brief review the methods presently utilised to measuring the economic values of insect pollinators for different end uses, highlighting the shortcomings of these methods in relation to their potential end uses. We then review the empirical literature and the proposed frameworks to highlight the main gaps in the evidence base.

2. Why measure the economic value of pollination services?

The economic value of pollination, as with any ecosystem service, has a number of potential, context-specific uses. First, economic valuation of ecosystem services is a means of illustrating the value (benefits) of conserving pollination services (Costanza et al., 2014), and alerting policy makers and other stakeholders of the risks of these services diminishing, risks which they may not have previously considered (Abson and Termansen, 2010).

Secondly, once quantified economically, the market and non-market values of pollination can be included as part of cost-benefit analysis to inform policy or business decisions and land planning (Hanley and Barbier, 2009). For instance, a decision on whether to maintain the current EU ban on neonicotinoid pesticides could be informed if the economic benefits of restricting the use of such pesticides, in terms of foregone pollination services, could be compared with the economic costs of such a policy, such as declines in agricultural yields (Goulson, 2013). Similarly, the economic benefits of enhanced wild pollinator populations arising from agri-environmental measures could be compared with the costs of such schemes, in order to prioritise and rationalise public expenditures to enhance the production of public goods (Breeze et al., 2014).

Finally, valuation allows for the construction of extended or environmentally-adjusted national accounts which show the value of changes in a country's natural capital, and to track changes in the value of the ecosystem and other assets which make up this natural capital stock (Barbier, 2011). Internationally agreement is slowly emerging on the importance of registering the economic value of ecosystem service flows in national economic and environmental reporting and accounting (ONS, 2012; United Nations Environment Programme, 2012; United Nations, 2013). An environmentally-adjusted value for Net Domestic Product (a measure of national income) would ideally incorporate both market and non-market benefits which are supplied by pollinators in any year, and also include a depreciation/net investment term to capture year-on-year changes in the capital value of the asset—its ability to provide direct *and* indirect benefits over time. However, the value of benefits to crop producers in year t from pollinators would not be added to the adjusted Net Domestic Product in year t since that value would already be included in the value of agricultural production (Nordhaus, 2006), although the benefits to consumer welfare (changes in consumers surplus) could be added (e.g. Gallai et al., 2009).

3. Conceptual frameworks for measuring the economic benefits from pollinators

In this section, the ways in which stocks of pollinator populations generate economic values is explained for (i) market-valued outputs (ii) non-market values. This leads to an explanation of

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how such values can be quantified for applied use. From an economic valuation viewpoint, economic value is often thought of in terms of marginal values: that is, as a pollinator population rises or falls by one "unit" (e.g. by one additional colony in a landscape), what is the change in benefits people potentially receive from pollinators?

3.1. Market values

Pollinators primarily provide economic value to crop production through increasing the quantity and quality of crops produced, resulting in greater economic output which is in turn influenced by market prices for the crop. The extent of these benefits will vary between crops (Klein et al., 2007) and varieties (e.g. Hudewenz et al., 2014) depending on the degree of floral self-compatibility. As such, pollination services (*PS*) act as an input to crop production in a similar manner to conventional inputs such as plant protection products. This effect can be captured as part of a crop output production function, a form of economic model which relates the physical yield of a given crop, $x1$, to variations in the supply of pollination services *PS* and other inputs:

$$Q(x1) = f(Y, PS, \varepsilon) \tag{1}$$

where $Q(x1)$ is the economic output per hectare per year, Y is a vector of other inputs (e.g. labour hours, pesticides etc.) and ε represents stochastic factors such as rainfall and temperature. There will be a separate production function for each crop ($x1$, $x2$, $x3$..) relevant to a farmer's choices of what to grow. The variable *PS* effectively represents the probability that any given flower will be sufficiently pollinated (v) above a threshold number of grains required to produce marketable fruit of a specified quality. As such, the lower the chance of a flower being sufficiently pollinated, the higher the marginal value of additional pollination services. As farmers will also have the option of switching between crops, these values are also heavily influenced by the costs of substituting between crops, particularly if switching to non-insect pollinated crops. This cost is likely to be low in arable crops but much higher in orchard and small fruit crops, where substitutes are themselves pollinator dependant (Klein et al., 2007).

However, the supply of pollination services is different from other agricultural inputs because of its direct link to plant reproduction. In some crops, pollination is essential to producing any output at all, and thus other inputs in the "no pollination" scenario can still generate zero output for some crops. For other crops, pollination only slightly enhances yield (Klein et al., 2007). Pollination services could therefore be distinguished from other inputs by using a scaling function *Z*, the value of which lies between zero and one, which is then applied to *potential* output which depends on use of other inputs such as fertiliser and labour:

$$Q(x1) = \{(f'(Y))Z(PS)\}, \varepsilon \tag{2}$$

Through this direct link to production, pollination also influences consumer welfare. By maintaining supplies of a crop relative to demand, pollination acts to keep prices to consumer moderated to less than they would be absent such pollination, resulting in a *de facto* increase in consumers' surplus(Gallai et al., 2009; Lautenbach et al., 2012).

Unlike many other inputs, pollination services are often provided for little or no cost to the producer, particularly wild pollination services which are often produced from habitats that are left to develop on land with poor access and productivity, minimising the opportunity costs to the producers. Mass flowering crops can themselves sustain crop pollinators through the temporary abundance of floral resources, effectively creating a positive feedback loop (e.g. Holzschuh et al., 2011). As such pollinators provide a service for a low cost that would otherwise have to be paid by the producer if they wished to optimise yields (Allsopp et al., 2008; Partap and Ya, 2012), reducing their marginal production costs per unit and as such increasing producer welfare (Kasina et al., 2009).

3.2. Non-market values

Beyond crop production, insect pollinators provide a number of non-market benefits. From an economic value viewpoint, this happens in at least two ways. First, individuals derive pleasure from seeing pollinators (a use value) and knowing they exist (a non-use, or existence value). Such values are direct benefits to individuals from the presence, diversity and abundance of pollinators. As such, changes to the presence, abundance and/or diversity increases will change well-being. The monetary value of such changes in utility is given by an individual's willingness to pay (WTP) for an improvement or to avoid a loss of pollinators. For an individual a, we could write:

$$Ua = f(S1, S2, S3, Y, N, E) \tag{3}$$

where Y is income, capturing ability to pay, E represents other environmental attributes which the individual cares about, and N is all other goods and services in the individual's choice set. The marginal, direct non-market value of a change in population $S1$ is given by wtp^* in (4), where $S1^0$ is some initial population of pollinator species $S1^1$ is a higher population level (here we assume that $S2$ and $S3$ are un-changing):

$$Ua\left(S1^0, S2, S3, Y, N, E\right) = Ua(S1^1, S2, S3, Y - wtp^*, N, E) \tag{4}$$

The marginal utility and thus WTP for pollinator diversity or abundance will likely be lower, the higher the level of species diversity in an area.

Second, individuals may benefit from the consequences of pollinators' actions, thus enjoying an additional, indirect benefit. For example, this could be through the effects of wild pollinators on the diversity and abundance of wild flowers and trees. Several studies have noted that respondents derive greater utility from increasingly floristically-diverse landscapes (Lindemann-Matthies et al., 2010), indirectly benefitting from the actions of pollinators. Wild pollinators are also important for the production of fruit and seeds for wild birds through their action on wild and garden plants (Jacobs et al., 2009), thus indirectly contributing to the utility from bird-watching. If we assume that the variable E in Eqs. (4) and (5) captures the importance of wild flowers and trees and of gardens and allotments to people, then the indirect, non-market economic value of pollinators is given by the effects of changes in pollinator populations on E. Ideally, we would want to empirically measure the partial derivative of E with respect to S multiplied by the partial derivative of U with respect to E.

4. Empirical evidence on the application of pollination service valuation.

4.1. Market values

The majority of studies into the economic value of pollination services have been purely illustrative and have been used to raise awareness of the impact of pollination services on regional or national agriculture (e.g.: Jai-Dong and Chen, 2011; Calderone, 2012). Almost all of these studies have focused on the producer benefits to crop production. Early studies equate the total value of crops benefitting from insect pollination (Matheson and Schrader, 1987; Costanza et al., 1997) or rents paid to bee-keepers for pollination services (Burgett et al., 2004) with the value of pollination services. However, neither of these methods are suitable proxies, as

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most crops are able to produce some yield in the absence of insect pollination (Klein et al., 2007), and as other inputs are used in production. Many countries such as the UK lack a well defined market for honeybee pollination services (Carreck et al., 1997). Furthermore even in countries where hive rental is widespread, honeybees often provide only a minority of pollination services (Garibaldi et al., 2013) and hiring them for pollination is uncommon for some crops (e.g. Oilseed Rape; Carreck et al., 1997). More recent studies have focused on estimating the value of pollination as an input into crop production using a simplified production function known as a dependence ratio (DR) which measures the proportion of crop output lost without pollination services. This DR approach was taken in the UK's National Ecosystem Assessment (Smith et al., 2011) to measure the economic value of pollination for all UK crops in 2007, and has been similarly applied in numerous countries and crops (Table 1). DR values are typically based on field research into the impacts of pollination services on yield or reviews of this work (e.g. Klein et al., 2007) and can capture the variation in benefits from insect pollination of different crops. As such, the marginal benefits of pollination will rise in proportion to DR. The DR approach has served to illustrate the benefits of pollination by highlighting regions where total production is especially vulnerable to pollinator declines, and to highlight areas where spending on conservation would be most beneficial (e.g. Lautenbach et al., 2012). Cook et al. (2007) also use this method to compare the benefits of pollinator conservation to the costs of preventing *V. destructor* from affecting wild honeybees in Australia. Although easily applied and updated using regional or national production statistics, DR values are susceptible to a number of biases based on the studies they are drawn from. Firstly, these studies may not use standardised methods, leading to bias arising from methodological differences. Second, studies used may not account for all economically significant outputs of crop output that are influenced by pollination, such as crop quality or producer costs, or all inputs (Garratt et al., 2014). Third, applying a single DR value to a crop may mis-represent varietal differences in pollination service benefits (e.g. Hudewenz et al., 2014), which is particularly important where varietal turn-over is high. Fourth, these studies usually assume that there are no benefits to reduced production when producers elsewhere profit from higher prices (but see Winfree et al., 2011). Finally, DR methods innately assume that pollination services are already at a maximum, which may not be the case (e.g. Garratt et al., 2014) and only estimate a 100% loss of pollination service rather than marginal losses.

Some studies have attempted to rectify these faults by assessing the per hectare benefits from comprehensive field studies that account for the effect of market quality benefits, cultivar variations (Garratt et al., 2014) and storage life (Klatt et al., 2014) as well as the effect on varying producer profits. Like DR studies, these yield analysis (YA) studies remain largely illustrative as they lack mostly lack the information to link marginal changes in pollination services to changes in output (but see Ricketts and Lonsdorf, 2013). However, if supported with sufficient data relating pollination services to local landscapes, it is possible for these small scale studies to develop estimates of the potential natural capital value of pollination services from particular surrounding habitats. This application has only been undertaken once by Ricketts and Lonsdorf (2013) for coffee production in Costa Rica. There are however a range of issues in extrapolating upward from any small scale study, most notably the representativeness of the site or landscape (Eigenbrod et al., 2010) and the marginal variation in demand for pollination services. Other studies have expanded the DR model to illustrate the impacts of pollination services on consumer welfare, using econometric techniques to calculate losses in consumer surplus (CS); an economic measure of the disparity between the price paid for a good and the highest price that the public are theoretically willing to pay. Although more comprehensive than DR studies, accurate estimates of the relations between crop prices, production and demand require data intensive econometric analysis (e.g. Southwick and Southwick, 1992), ideally involving complex trade analyses to allow for trade effects, an enhancement that most current CS studies have not incorporated (Kevan and Phillips, 2001). As CS estimates are extensions of DR analyses, they also suffer from many of the same flaws as DR analyses. Furthermore, these

Table 1
Published studies assessing the economic value of pollination services.

Study	Region	Value (2010 GBP)	Method
Matheson and Schrader (1987)	New Zealand	£1.83 bn	Crop value
Costanza et al. (1997)	Global	£118.4/ha	Crop value
Calderone (2012)	USA	£10.6 bn	DR
Jai-Dong and Chen (2011)	China (horticulture)	£29.3 bn	DR
Kasina et al. (2009)	Kenya (small holdings)	£25–£1910/ha	DR
Losey and Vaughn (2006)	USA	£2.30 bn	DR
Morse and Calderone (2000)	USA	£12.1 bn	DR
Zych and Jakubiec (2006)	Poland	£520.2 M	DR
Carreck and Williams (1998)	UK	£322.1 M	DR
Canadian Honey Council (2001)	Canada	£406.2 M	DR
Gill (1991)	Australia	£0.5-£0.9 bn	DR
Pimtel et al. (1997)	Global	£165.7 bn	DR
Guerra-Sanz (2008)	Spain (glasshouse)	£470 M	DR
Brading et al. (2009)	Egypt	£1.3 bn	DR
Robinson et al. (1989)	USA	£12.4 bn	DR
Garratt et al. (2014)	UK (Apples)	£36.7 M	YA
Klatt et al. (2014)	EU (Strawberries)	£750.7 M	YA
Stanley et al. (2013)	Ireland (Oilseed Rape)	£3.32 M	YA
Greenleaf and Kremen (2006)	USA (sunflower)	£16.6 M	YA
Olschewski et al. (2006)	Indonesia & Ecuador (coffee)	£30-£31/ha	YA
Ricketts et al. (2004)	Costa Rica (coffee)	Up to £97/ha	YA
Shipp et al. (1994)	Canada (glasshouse peppers)	Upto £41,6450/ha	YA
Gallai et al. (2009)	Global	£121.8 bn	DR, CS
Southwick and Southwick (1992)	USA	£2.5-£8.3 bn	DR, CS
Allsopp et al. (2008)	South Africa	£17.9–£78.6 M	RC
Calzoni and Speranza (1998)	Italy (plums)	£274/Ha	RC
Winfree et al. (2011)	NJ, USA (watermelons)	£0.13-£2.3 M	RC, YA, CS

Legend: DR=Dependence ratio; CS=Consumer surplus, RC=Replacement costs, YA=Yield analysis.

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estimates unrealistically assume static producer welfare, whilst imports may be able to satisfy consumer demands at negligible additional costs, damaging the welfare of producers that are not able to readily switch their crops. An alternative to these basic production function approaches is to examine costs avoided by the presence of pollinators by estimating the costs of replacing them (e.g. Allsopp et al., 2008). Unlike DR studies these replacement costs (RC) methods are less susceptible to geographic or cultivar variations and do not require assumptions to be made regarding current service levels. Again, these studies remain almost exclusively illustrative due to their inability to highlight the effects of marginal changes in both insect or artificial pollination services and the impacts on crop prices that would result from the adoption of such methods. It is unlikely that this method would be applicable for all crops, as artificial pollination methods have proven ineffective on a number of crops (e.g. Kempler et al., 2002), and, more importantly, are unlikely to accurately estimate the real value of a beneficial ecosystem service, due to issues of substitutability, joint products and the need for the least-cost alternative to be considered when such costs are calculated (Hanley and Barbier, 2009).

More recently, Winfree et al. (2011) combined DR, CS and RC methods into a single assessment of the value of pollination for watermelon production in Pennsylvania based upon a detailed YA study. This combination of methods produced a more comprehensive examination of pollination service benefits to the crop, both within and beyond the state of Pennsylvania. However, much of the data collected remains very case specific and is of limited use in broader cost-benefit analysis.

4.2. Non-market values

Whilst many studies in the literature have applied stated preference methods to estimate the value non-market benefits of biodiversity, at present only one study has undertaken stated preference estimates of either direct or indirect non-market pollinator benefits,(Mwebaze et al., 2010). This resulted in an estimate for the existence value of protecting honeybees in the UK of £1.77 bn/year. However, this study is based on a small and non-random sample of the public, whilst the question used to elicit willingness to pay means that this figure confuses the market- and non-market values of pollinators. Moreover, since the survey did not contain any statement regarding the consequentiality of responses, there was no incentive for participants to reveal their true values.

5. Expanding the framework

Although economic valuation of pollination services has a number of potential end uses, our review has highlighted that presently most studies remain purely illustrative. Although illustrative research has uses in raising awareness, policy engagement on pollination services has become particularly strong in many countries with major policy initiatives such as the UK's National Pollinator Strategy (DEFRA, 2014). In the following section we present an expanded framework for market valuation and propose methodologies for non-market valuation of pollination service benefits that can facilitate valuation which is more useful to such policy goals.

5.1. Expanding methods—market valuation

Most existing valuations of pollination services are not applicable to cost-benefit analyses concerning particular policies because they do not measure the impacts of marginal shifts on end products. At a primary, bio-economic modelling level, it will be of critical importance to expand the production function models described previously in order to assess the full breadth of impacts arising from pressures, and to enhance the transferability of values between sites required in natural capital asset valuation.

As pollination services are provided by communities of mobile organisms they are largely stochastic and depend on a range of factors within the community. Foremost, the supply of pollination services PS is influenced by both the relative pollination efficiency of different insect species (honeybees, bumblebees, hoverflies etc.), S, and the overall diversity of taxa within the landscape. The efficiency of individual pollinator species, is in turn a product of their effectiveness as pollinators (E), usually in terms of pollen grains deposited/visit (e.g. Winfree et al., 2011), their visitation rate/period (T) (e.g. Woodcock et al., 2013). This in turn will be affected by the probability of S1 making a legitimate visit (R) and their overall abundance (A) within the landscape. The pollination services provided by an individual species can therefore be expressed as:

$$PS(X1)S1 = h((E,T)R,A) \tag{5}$$

This framework allows for direct modelling of the economic impacts of drivers that affect pollinator efficiency, particularly in systems that rely heavily upon a single pollinator such as glasshouses. For example, a new line of a crop developed with more accessible nectar may result in a 10% visits/day. By understanding the whole function at (5) this model provides an insight into the economic returns expected which can be compared to the costs of the new variety.

In most systems however, pollination services are provided by a range of species in tandem. These species may each provide services independently, resulting in additive benefits. However in several systems, pollinators may act as compliments, with the activities of one species or group of species enhancing the service efficiency of others (e.g. Brittan et al., 2013; Greenleaf and Kremen, 2006). Alternatively, species may act as substitutes for one another, maintaining service levels under otherwise stochastic factors such as the population declines and adverse climate conditions (e.g. Brittan et al., 2013; Winfree and Kremen, 2009). The rate of substitutability between pollinators (ie if PS from S1 falls by 5%, PS from S2 raises by x%), particularly between managed and wild species provides information on the insurance value of pollination services against the loss of key species e.g. in the event of a major disease outbreak (Baumgartner, 2007).

$$PS(x1) = g(S1, S2, S3....) \tag{6}$$

A community level production function such as (6) allows for an assessment of the costs and benefits of policies that are likely to affect pollinators; for instance is a new agrochemical causes a 10% decline in $S1$ but leaves $S2$ and $S3$ unaffected, how will this affect output of $x1$? As the abundance of any species (e.g. S1) within the community changes, the function should be able to trace out changes in the marginal physical product of $S1$. The overall community composition from a landscape will be influenced by a number of local factors, including the intensity of pressures (e.g. agrochemicals) local foraging resources and the strength of a source population (Scheper et al., 2013; Kleijn et al., 2011). These can be modelled as part of separate production function based on existing projective models such as the inVEST model (Lonsdorf et al., 2009) and linked back to service value by way of the production functions described above.

Finally, production function models can be further expanded by incorporating measures of farmer costs in order to examine the effects of different cropping patters within the landscape and interventions on overall profits. Such a profit function could be defined at the level of the farm. In this case, the prices of all

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outputs (crops) the farmer could grow, along with the costs and marginal physical products of each input, would be relevant to determining the maximum profit he can make, and determining the combination of crops and management regime which result in this maximum. The effects of changes in PS on profits could thus be estimated. Existing modelling approaches beyond pollination studies have been used to describe such an optimisation system, incorporating ecological links between farm management and ecosystem service supply or biodiversity, as described for example in Armsworth et al. (2012) and Hanley et al. (2012). Such approaches link the PS values estimated by the production function models to optimal output decisions where producers can choose crop types and input uses.

5.2. Expanding methods—non-market valuation

To date, the non-market benefits of pollination services have only been vaguely explored. As such, these benefits remain a major uncaptured knowledge gap, despite the availability of methodologies to fill the gap. Economists have developed a range of methods for empirically estimating such non-market values (Hanley and Barbier, 2009). For both direct and indirect non-market values of insect pollinators, it seems likely that only stated preference approaches would be a feasible method. Stated preferences work by asking a sample of individuals to either state whether they would be willing to pay a particular sum of money for an increase in an environmental good, or their willingness to accept compensation for a decline in this good (contingent valuation); or by asking people to make choices between different "bundles" of environmental attributes and a price (choice experiments). These responses are obtained in the context of a carefully-constructed hypothetical market for the good in question. Features of such markets which have been shown to be important are (i) that respondents feel that their responses are consequential (Vossler et al., 2012); (ii) that a non-voluntary payment mechanism be used (iii) that the environmental change in question be clearly described, and that any uncertainty over this environmental change is also well-described and (iv) that the hypothetical market is realistic and does not encourage ethical rejection.

For direct benefits, where people care about the populations of pollinators, either contingent valuation or choice experiments could be used to estimate willingness to pay for a change in such populations (e.g. a 10% increase in bumblebee abundance over a 5 year period in England). Choice experiments would enable the researcher to measure the impacts of different attributes of such a policy change on people's preferences—such as whether they prefer an increase in species diversity rather than abundance, and whether they prefer policy to be targeted at endangered or common species. Either method could be used to show how the non-market, direct benefits of pollinators vary across the country and across income groups, or between rural and urban households. Both methods have been used to estimate the utility benefit of changes in biodiversity across a wide range of settings beyond pollinators (Atkinson et al., 2014; Jobstvogt et al., 2014). The main challenges of applying such methods to estimating the non-market values of pollinator populations would be to meet the good design requirements noted as (i)–(v) above. Moreover, individuals may feel that they lack sufficient knowledge about the ecological importance of pollinators to be able to state their preferences in terms of Willingness to Pay for prospective changes in pollinator populations (Christie et al., 2006), although methods are available which can reduce this lack-of-knowledge barrier to valuing changes in biodiversity (e.g. LaRiviere et al., 2014; Colombo et al., 2013).

For indirect benefits, choice experiments and contingent valuation could be used to value marginal changes in the environmental goods which pollinators help to produce, such as wild flower meadows. However, scientists would also need to be able to provide information which would enable this change in environmental quality to be quantitatively related to an under-lying change in pollinator populations, in order to "back out" the implied non-market value of this change.

6. Knowledge gaps

Whilst there is a clear conceptual basis for measuring the economic value of insect pollinators to the detail required for application by policy analysts, and whilst there are a range of methods that exist for estimating these values, there remain deficiencies and omissions in the empirical literature on market values of pollination, and an almost total lack of empirical studies on non-market values. One significant barrier to wider and better use of production function approaches is the lack of generalizable, empirical functions which relate pollinator efficiency, abundance and diversity to crop output. Non-market valuation studies are limited by the complexities in extrapolation from case-specific valuation of pollination services, and by problems in identifying indirect linkages to well-being.

6.1. Market based valuation

To date, studies on the efficiency of individual pollinators have focused on comparisons between different managed species (e.g. Thompson and Goodall, 2001) although more recent studies have begun to address the contributions of wild species groups (Winfree et al., 2011). However, even these studies have not provided enough information to generalise species service efficiency beyond their particular case study areas, focusing instead on observations of individual species or generalised groups. A traits based approach, in which morphological (e.g. size or tongue length) and behavioural traits (e.g. activity period) are linked to individual species efficiency (Ne'eman et al., 2010) would allow for a greater degree of benefit transfer on a per-crop basis. Values could theoretically be linked with crop flower traits (e.g. flower size, pollen production) in order to better generalise relationships, and thus to identify economically-significant wild and managed pollinator populations.

Pollinator abundance is important in determining both individual species efficiency within the landscape (Eq. (4)) and their contribution to overall service delivery (Eq. (5)) which is in turn affected by the abundance of other species within the landscape. Several recent studies have linked the abundance and diversity of pollinators to their service delivery (Hoehn et al., 2008; Rader et al., 2009), however the effects of substitution and interaction between species have also only been explored in a few specific case studies (e.g. Greenleaf and Kremen, 2006; Rader et al., 2013). As with species efficiency, a traits-based approach linking diversity to services would be ideal for facilitating a more accurate and transferrable assessment of pollination service values.

Another research gap relates to threshold effects. Threshold effects in the supply of pollination services due to a decline in the condition of the pollinator asset would result in large changes in the marginal economic value of pollinators. These thresholds are likely to be lower for crops or wild plants that are more reliant on particular pollinators, such as field beans which are largely pollinated by long tongue bumblebees (Free, 1993). Areas reliant on honeybees are also vulnerable to collapse, as diseases can spread quickly between colonies, and can spill over into wild bumblebees (Furst et al., 2014). The integrity of the pollinator asset could decline in a non-linear way if there is a positive feedback between wild flower diversity loss and pollinator diversity. There are also issues of

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reversibility: once a population has suffered significant losses, it may be difficult or impossible to recover (Ellis et al., 2013). Understanding population thresholds and reversibility using existing ecological models of population dynamics is therefore a key factor when examining long term costs and benefits of actions that are likely to increase pressures on pollinators.

Although ecological research has linked the abundance and diversity of pollinators to landscape features such as agrochemical use and semi-natural habitat (Scheper et al., 2013) these links have yet to be widely generalised. As such, attempts to map the availability of pollination services still rely heavily on expert appraisal (e.g. Lonsdorf et al., 2009; Schulp et al., 2014), leaving it difficult to determine the potential or actual service delivery of pollinators at a landscape level. This knowledge gap, an essential barrier to accurate integration of wild pollinators to natural capital, could be most accurately filled by the development of systematic monitoring schemes, methodologies for which are already well established (LeBuhn et al., 2013). Unfortunately, while a number of schemes monitor the diversity of species within landscape, no systematic monitoring scheme has yet widespread trends in pollinator abundance, although the UK government has acknowledged the need for such a scheme (DEFRA, 2014). This would in turn allow for a more detailed assessment of the costs and benefits of different mitigation actions for pollinator populations, by comparing the status and trends of populations in landscapes with different management options.

Beyond the ecological aspects of estimating market benefits, our review highlights a limited number of studies examining market benefits to stakeholders beyond producers. While models exist to assess the impacts on consumer surplus (e.g. Lautenbach et al., 2012), limited information exists on how national crop prices react to yield changes, particularly when the crop is widely traded internationally (Kevan and Phillips, 2001). This prevents these models from being more widely used (Winfree et al., 2011). As such, econometric and partial equilibrium modelling analyses of these relationships would have substantial value for assessing the wider benefits of changes in pollination services and assessing the marginal costs of changing to non-insect pollinated crops. Furthermore, many of the studies examining consumer effects have only explored the value based on prices paid to farmers, which will often represent what supermarkets and other distributors will pay rather than what the end consumer pays, pointing to a need to examine the broader supply chain.

Finally, the links between abundance at an individual species level and the supply of pollination services at the landscape level represent a major challenge for modelling.

6.2. Non-market valuation

Although non-market valuation methods are well established for ecosystem services in general, they have yet to be applied to pollination services, in itself a significant knowledge gap. Part of the complexity in assessing these values is the inherent difficulty of separating the value of pollination from the value of pollinators—people may understand the importance of bees and be willing to pay to conserve them, but their values may be very different for hoverflies. Limited knowledge of the linkages between pollinators and pollination services is also likely to affect preferences and therefore the stated values of non-market values (Christie et al., 2006). A major knowledge gap in assessing these non-market values is therefore the extent of public knowledge and information on pollination services.

More significantly, there remains the challenge of identifying the links between marginal shifts in pollinator populations and the values attributed to the non-market benefits arising. When considering a single crop, the relation between pollination services and yield (benefit) is relatively straightforward due to the simplified plant-pollinator networks occurring in commercial crops. However, when attributing value to the aesthetic values of floral diversity, large plant-pollinator networks can be involved, adding additional complexity to assessments of the marginal changes in pollinator communities (Burkle et al., 2012). As such, it becomes difficult to generalise values for key pollinators and therefore to identify the impacts of marginal changes caused by pressures or mitigations. This is likely to be true of other indirect, non-market benefits from pollination services.

7. Conclusions

This study has reviewed the conceptual basis and scientific rational for evaluating the economic benefits of pollination services. Of the principal uses of valuation, existing work has focused almost exclusively on illustrative studies, with few studies presenting the potential to use values in cost-benefit analyses or natural capital accounting. To this end we have presented a detailed framework for valuing marginal impacts of shifts in pollinator communities on the market and non-market values associated with pollination services. The knowledge gaps identified highlight the significant ecological complexity of developing such models, with extensive field ecology required to build the comprehensive production function models to answer these questions. It also highlights the problems of combining ecological and economic models, and the data requirements for good economic modelling.

These knowledge gaps in turn highlight the principal difficulties in developing valuation in a form that would be suitable for cost-benefit analysis and/or natural capital accounting. Increasing complexity arises from the requirement to make assessments of benefits transferrable and comprehensive. While a number of localised studies have developed methods suitable for assessing cost and benefits (Winfree et al., 2011; Cook et al., 2007) and for natural capital accounting (Ricketts and Lonsdorf, 2013), these are highly case-specific. Extrapolating from these studies therefore runs the risk of presenting erroneous values and over-generalising (Eigenbrod et al., 2010). However, it is such broad, regional and national scale analyses that are of particular interest to stakeholders and the policy community (Vanbergen et al., 2012; United Nations Environment Programme, 2012). As such, if research is to achieve the demands for truly functional valuation it will be imperative for policy makers and other stakeholders to support further pollination services research. Based upon the principles of valuation set out above, we recommend the following as priority areas for new work: (1) Identification of key pollinators and pollinator traits in a range of representative crops, (2) assessment of the behavioural and morphological traits that facilitate substitution and synergy within pollinator communities, (3) evaluation of the links between habitat traits and the populations of pollinators, ideally using a systematic monitoring scheme, (4) econometric analyses of the links between insect pollinated crop yields, the prices paid for these crops and consumers' and producers' surplus; and (5) an assessment of the non-market benefits of pollination services utilising stated preference techniques.

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