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Food production, ecosystem services and biodiversity: We can't have it all everywhere

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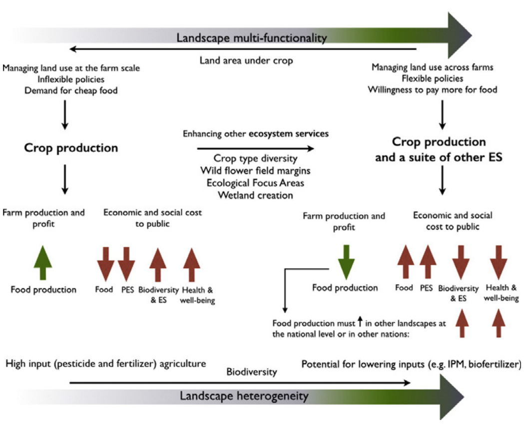
HIGHLIGHTS

- The ideal is to balance food production, ES and biodiversity in arable systems.
- We test a framework that reveals the multi-scale trade-offs between these functions.
- It shows how stringent pesticide policy may have effects on achieving food security.
- Policy change that promotes multifunctionality at landscape scales is required.
- But impacts on biodiversity and ES may have to be accepted in certain landscapes.

GRAPHICAL ABSTRACT



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ABSTRACT

Debate about how sustainable intensification and multifunctionality might be implemented continues, but there remains little understanding as to what extent they are achievable in arable landscapes. Policies that influence agronomic decisions are rarely made with an appreciation of the trade-offs that exist between food production, biodiversity conservation and ecosystem service provision. We present an approach that can reveal such trade-offs when used to assess current and future policy options that affect agricultural inputs (e.g. pesticides, nutrients) and practices. In addition, by demonstrating it in a pesticide policy context, we show how safeguarding a range of ecosystem services may have serious implications for UK food security. We suggest that policy change is most usefully implemented at a landscape scale to promote multifunctionality, tailoring pesticide risk assessment and incentives for management that support bundles of ecosystem services to specific landscape contexts. In some instances tough trade-offs may need to be accepted. However, our approach can ensure that current knowledge is used to inform policy decisions for progress towards a more balanced food production system.

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1. Introduction

Achieving food security through agricultural production depends on an eroding natural resource base, and with a growing global population the challenge to meet the demand for food with less environmental damage is pressing (Tilman et al., 2011; Garnett et al., 2013; Godfray and Garnett, 2014). This is recognised across science, policy, industry and NGO sectors. Consequently, the usefulness and implementation of concepts such as sustainable intensification (Garnett and Godfray, 2012; Garnett et al., 2013) and multifunctional landscapes (O'Farrell and Anderson, 2010) are being debated.

A truly sustainable intensification needs to promote multifunctional landscapes, ensuring the ecological functions that underpin crop production, and other ecosystem services (e.g. flood alleviation and recreation), are maintained. This can be achieved by optimizing land use and agricultural practices at the same time as lowering the required inputs of fertilizers (nitrates and phosphorus), through improved soil management, and of pesticides, through Integrated Pest Management (IPM) (Garnett and Godfray, 2012; Garnett et al., 2013). Presently, the EU, for example, is some distance from achieving sustainable intensification. Markets and policies drive land use changes at the individual farm scale, so emphasis remains on the delivery of private goods, to ensure food is a cheap commodity. However, to achieve sustainable intensification and multifunctionality, policy-makers need to shift to landscape scale thinking, working across farms and creating a governance that supports food production whilst ensuring the protection and enhancement of public goods (i.e. ecosystem services and the biodiversity on which they depend) (Fig. 1).

It has been suggested that the ecosystem services concept should be mainstreamed through EU agricultural policy (Plieninger et al., 2012), and, therefore, the objective in arable systems should be to secure food provision at the same time as safeguarding ecosystem services and biodiversity (Pe'er et al., 2014). However, while we can all agree with the concept, we do not yet know the answer to a key question; to what extent is multifunctionality realistically achievable in arable landscapes (areas dominated by agricultural fields, but also containing a mosaic of other habitats inbetween e.g. hedgerows, wetland, temporary grassland, small areas of woodland)? Indeed, there is little understanding about how different agronomic decisions impact on biodiversity and in turn the whole suite of ecosystem services that such systems can supply (and depend on) (Bommarco et al., 2013). Recent research shows that the relationship between biodiversity and ecosystem services can be both positive and negative, and that because biodiversity can contribute to multiple services there will be trade-offs between them (Harrison et al., 2014). The policies that influence how land is managed, what crops are grown and the level of inputs into the system, are rarely made with an understanding of the trade-offs that will actually occur between the services that can be provided, or their broader cross-scale socio-ecological implications. In addition, the effect of interactions between environmental policies, for example agricultural, water, conservation and pesticides, on farmers' land management decisions has not been fully explored (apparent contridictions between policies are being found, and these may have unintentional consequences on food production or ecosystem services). Consequently, it is hard to make informed decisions about which trade-offs are acceptable, or not, in which context. In addition, it is likely that what works in one place may not somewhere else and land management decisions can not be taken universally. Thus the likely reality is that we are unable to optimise everything (biodiversity, food and other services) everywhere, and there will be consequences of any decisions (ecological, social and economic) at a variety of scales. This important issue and the need to make decisions about what we are prepared to forgo and where, are rarely openly acknowledged in the conservation community. We now urgently require a way to ensure that any changes in the EU policies that influence agricultural inputs and practices can be informed as to the broad implications and trade-offs involved in doing so.

Here, we demonstrate a framework that can be used by policy makers to reveal multi-scale trade-offs between food production, biodiversity and other ecosystem services in arable systems (Table 1). Adopting it for decision making will enable effective analyses of the

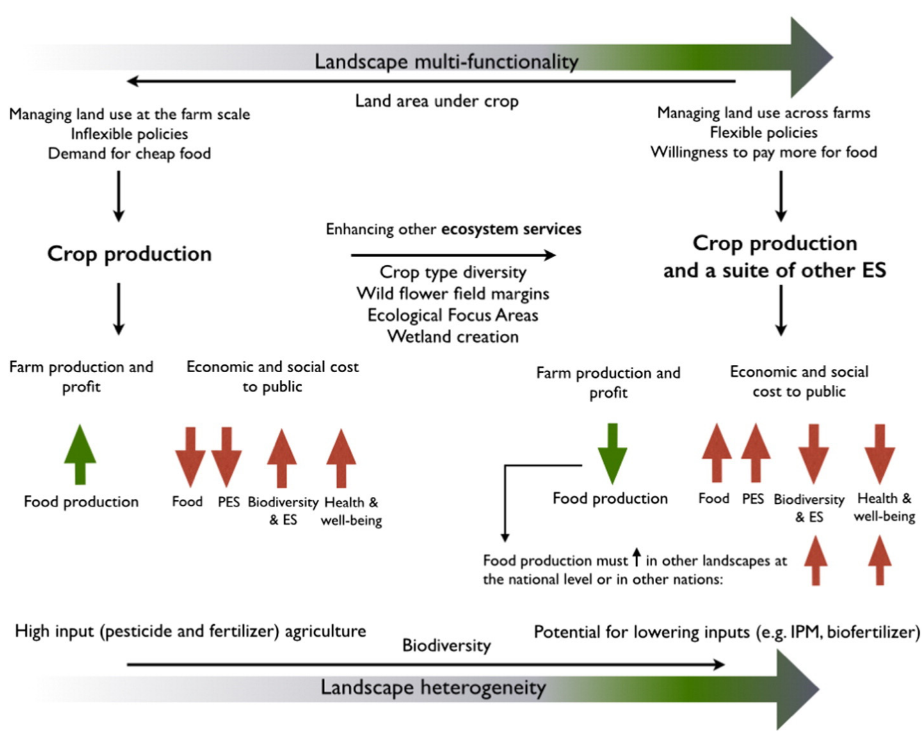

Fig. 1. Landscape multifunctionality can increase with landscape heterogeneity as there is a move from crop production at the expense of ecosystem services and biodiversity (left side of the chart), to a production that enhances it though the use of green infrastructure, such as wild flower field margins (right hand side). However, it may not be possible, or wise, to achieve such multifunctionality across all landscapes if we are to feed a growing population. PES – Payments for Ecosystem Services, IPM – Integrated Pest Management.

current situation and alternative scenarios on which to decide what we want (and can realistically have) from arable landscapes. We suggest not only how our approach can be used to understand the implications of policy change, but how a change in scale of thinking is required when managing for multifunctionality. Finally, we outline areas of research that need to be addressed to bring us closer to implementing a more balanced food production system.

2. Materials and methods

2.1. Identifying drivers of crop production, and establishing the scenarios (framework steps (i) & (ii))

We analysed the factors that influence wheat production in the UK (Table S1) and used our framework (Table 1) to identify the trade-offs between food production, biodiversity and ecosystem services that arise from the recent changes in EU pesticide policy and regulations (e.g. the review of pesticide authorisations and restriction of neonicotinoid use) on winter wheat production. We used a set of scenarios that reflect realistic agronomic decisions that farmers might make under particular circumstances and began by using the most popular choice by farmers in the UK, the 3-year rotation for wheat production (winter wheat, winter wheat and oilseed rape): Scenario 1. (a) Winter wheat, winter wheat, oilseed rape before neonicotinoid moratorium, (b) winter wheat, winter wheat, oilseed rape after neonicotinoid moratorium, Scenario 2. Winter wheat, winter wheat and winter beans (a logical 3 year rotation should the outcome of the neonicotinoid ban be that farmers choose not to take the risk of growing oilseed rape finding that yields are not good enough without neonicotinoid seed treatments). From there, we made assumptions, using expert knowledge, as to what the most likely crop choices would be given the changes in pesticide legislation, also taking into account the recent moratorium on the availability of some insecticides to farmers.

2.2. Inputs and impacts on ecosystem services (framework steps (iii) & (iv))

Each active substance within the herbicide, fungicide and insecticide products most commonly used in winter wheat, oilseed rape and winter beans was established (Garthwaite et al. (2013) and expert knowledge). This resulted in 30 active substances being assessed. The potential impact on 9 key arable ecosystem services was then assessed. These particular ecosystem services were chosen as they are generally considered to be some of the most important ecosystem services provided by arable systems (Power, 2010) and were identified by EFSA documents discussing the impacts of pesticides on ecosystem services (EFSA 2010, 2014). Whereas this is not an exhaustive list of ecosystem services it is representative and appropriate for demonstrating the approach. Assessing the impact of active substances on ecosystem services was achieved by establishing (i) the ecotoxicological profile and related level of hazard for each active substance, (ii) the level of risk and whether it required mitigation; and (iii) which mitigation measure(s) is (are) required. For the purpose of this analysis, the ecotoxicological profile of each active substance was extracted from the IUPAC Pesticides Properties Database (PPDB) (http://sitem.herts.ac.uk/aeru/iupac). The PPDB holds data for all European approved active substances and their metabolites, extracted from the official documents in support of the European assessment of pesticides, as well as from other European and North American databases. The ecotoxicological hazard (Low, Medium or High) posed by each active was established from the database. The associated risk of each active to each of the standard ecotoxicological taxonomic groups was then derived based on the application rate for each crop, taken from the European peer review reports (www.efsa.europa.eu/en/publications/efsajournal.htm*)*. Potential environmental risks, if any, for the uses considered here were also identified in the European peer review reports (www.efsa.europa.eu/en/publications/efsajournal.htm), and/or review reports published by the European Commission (http://ec.europa.eu/food/plant/protection/evaluation). Risks are evaluated in the regulatory context of the authorization of a product on the market (EC, 2009a) and evaluate the potential impacts that the product could exert in the context of its use. Additional risks that may either be the result of other stressors or their possible interactions with the product are not taken into account in this regulatory risk assessment (EC, 2009a). The European regulatory risk assessment process therefore assumes that regulating on acceptable risk for individual products is sufficiently protective against the overall risks of a crop protection programme; an assumption that has been evaluated via experimental (Arts et al., 2006) and modelling studies (Focks et al., 2014).

In many cases, the recommended application rate for the active substance on each crop was sufficient to mitigate risks, even for a substance that displayed a non-negligible toxicity to a group of organisms. Indeed it is the purpose of the European assessment to verify that the level of exposure corresponding to the intended uses remains below the threshold for effects, including a margin of safety to account for extrapolation and uncertainty issues. In these instances, we concluded that there was Low Risk at the Recommended Application Rate. If this was not the case, additional risk mitigation measures would be needed to ensure that the level of exposure would remain lower than the threshold for effects. Buffer zones are recommended to protect aquatic organisms, non-target arthropods and terrestrial plants from spray drift when applying pesticides. These buffer zones are non-vegetated strips (buffer width is calculated without drift interception hence they are assumed to consist of bare soil) of a defined width at the edge of a field between the crop and a water body, if protecting aquatic organisms, or between the treated crop and non-treated (cropped and non cropped) areas on all sides of the field for terrestrial organisms. In the case of potential risks related to transfers via runoff, then specifically a vegetated strip between the crop and the water body is most often recommended. The width of a buffer zone or vegetated strip is determined via a risk assessment that includes risk mitigation measures. For some chemicals, precautions with regards to application timing may be recommended to protect birds or mammals, e.g. to avoid the reproductive period. Another measure applicable to “solid formulations” i.e. granules and coated seeds may be the removal of spillage of granules, or treated seed. Potential risks to pollinators are also mitigated through recommendations to not apply pesticide onto flowering crops, and to monitor the presence of bees in the vicinity of the treated crop. These recommendations are reported on the labels of pesticide products in accordance with Regulation (EU) No 547/2009 (EC, 2009b). In order to ensure the risk assessments and related risk mitigation measures used in this analysis were as up to date as possible, they were established based on the conclusions of the most recent European peer review, and adapted to the specific uses considered here. When European peer reviews were not available, conclusions of the

Table 1
The ecosystem services framework for revealing trade-offs in crop production.

Framework steps
(i) Identify the drivers of crop production (social, economic, policy and bio-physical)
(ii) Identify the current land management (crop rotation etc), and establish scenarios of potential changes in farmers' agronomic decisions due to the particular aspect of (i) that is of interest, and the consequences for land management as a result of change in driver(s)
(iii) Scope out the requirements for providing particular land management scenarios (inputs like pesticides, or fertilisers N and P)
(iv) Identify the impacts of (iii) on key aspects of biodiversity (service providing units) and ecosystem services
(v) List the consequences of impacts (the trade-offs: environmental, social and economic) and possible outcomes of these for each scenario with reference to the farm, landscape, UK and global scales
(vi) Design solutions to mitigate impacts and maximise multifunctionality – type and location of elements of green infrastructure and risk mitigation options

most recent evaluation performed in individual Member States were used (see for example https://www.anses.fr).

The consequent direct impacts of pesticide exposure on ecosystem services were estimated in accordance with EFSA's scientific opinion on protection goals (EFSA, 2010). Table S2 shows which arable ecosystem services are likely to be impacted if the pesticide should pose a direct risk to a particular taxonomic group (via the ecosystem processes they contribute to). The assumption here is that the taxonomic groups thought to play a functional role in the delivery of each ecosystem services are the key contributors, albeit to a greater or lesser extent through time. Pesticides may also have indirect/food web effects, some of which will also affect crop production. The roles of the taxonomic groups in Table S2 were translated into ecosystem service impacts in Fig. 2. The groups of substances that posed either a high or medium hazard to the taxonomic groups were considered to have a potential impact on ecosystem services. It is worth noting that multiple taxonomic groups play a role in the provision of all the key ecosystem services. Ecosystem services change from 'impact' to 'reduced impact' if one or more of the taxonomic groups that underpins that service moves from a hazard status to a low risk risk status, if all of them move from hazard to low risk then there would be 'no impact'.

Understanding potential impacts (framework step (v)) allowed an overall evaluation of the trade-offs inherent in growing each crop (Fig. 2), the consequences to farmer decisions about crop production, e.g. whether an alternative pesticide could be used, a loss of yield accepted, or if there would be a switch to another crop because the loss of yield would be too great to be economically viable (Table 2). The effect was scaled up to understand the viability of crop rotation choices in each scenario (Table 2).

3. Results

3.1. Wheat production in the UK: a case study

We used the framework (Table 1) to illustrate how changes in EU legislation that affect pesticide use and risk assessment might influence wheat production in the UK. Pesticides are a significant input in EU arable systems at present as they can ensure high crop quality and yield. In the move towards the sustainable intensification of agriculture, there is likely to be a more targeted use of pesticides, and they should become one component of a diverse IPM toolbox. We do not discuss the direct benefits of IPM, including reduced pesticide use, on biodiversity and ecosystem services as this has been covered in depth elsewhere (for reviews see Pretty and Bharucha, 2014; Kremen and Miles, 2012) but this could be equally well assessed using our illustrated framework. Wheat is the most widely grown crop in the UK, with 1.8 million ha planted in 2015 (DEFRA, 2016). The most common break crop for use in a winter wheat rotation is oilseed rape (652,000 ha grown in 2015 (DEFRA, 2016)), with winter beans an alternative break crop where oilseed is not viable (field beans sown 170,000 ha in 2015 (DEFRA, 2016)). Regulation of the use of new pesticides (Sustainable Use of Pesticides Directive 2009/128/EC) has recently been tightened, authorisations for pesticide products are being reviewed, and certain substances have been restricted (e.g. 3 neonicotinoid pesticides for 2 years due to their potential risks to bees (Regulation (EU) No 485/2013)). As a consequence the range of chemicals that can be used to produce winter wheat, including those used in the break crops that support wheat rotations (oilseed and beans), is shrinking. Availability is also being compromised as crop pests become resistant to particular chemicals. Black grass, a challenging widespread perennial weed pest that can cause up to 50% yield loss in winter wheat (Moss, 2013), is a particular problem in this respect. Herbicide resistance in black grass is now widespread with 1.2 million ha of the UK affected (BGRI, 2015). The cabbage stem flea beetle and turnip yellow virus (TYV) transmitted by aphids can cause average annual yield losses in oilseed of ~15,336 t and ~206,010 t respectively, if the crop is left untreated (from statistics in Clarke et al. (2009) and DEFRA (2013)). In the absence of neonicotinoid seed treatments the alternative insecticides for use in oilseed may not be effective as resistance is an issue, particularly in the case of the TYV. Simultaneously, the risk assessment process for pesticides is becoming more stringent, as a consequence increased mitigation measures need to be put in place to meet the estimated levels at which "no unacceptable risk" to non-target organisms and related ecosystem services are expected (see Supporting Information). Given this current context a broad scale analysis of the consequences to wheat production, biodiversity and ecosystem services, of policy changes is required urgently.

We identified a set of scenarios that reflect realistic agronomic decisions that UK farmers might make when growing winter wheat and the most commonly planted break crops, given the recent changes in the pesticide legislation. This resulted in the assessment of the impacts on biodiversity and 9 key arable ecosystem services of 30 active substances used to produce these crops. This required establishing the ecotoxicological profile and related level of hazard and risks, need for risk mitigation measures (buffer strips) for the use of each active in field, and finally which taxonomic groups play a role in the provision of the ecosystem services of interest (Supporting Information and Table S2).

Our analyses show that it may not be possible to maintain wheat yields, or in some circumstances even viably grow the crop and its common break crops in the near future, whilst potentially safeguarding a suite of important ecosystem services associated with arable ecosystems (Fig. 2, Tables 2). Changes in pesticide risk assessment results in farmers having to implement broader mitigation measures in order to continue to apply common pesticides (see Supplementary Information). We found that under new pesticide labelling buffer strips (which are defined as unvegetated bare soil buffer strips, see supplementary information) of 50 m are required for insecticide use across wheat, oilseed and bean crops, with 20 m wide buffer strips required for herbicide and fungicide use in oilseed (Fig. 2, Table 2). Buffer strips are required for mitigation against spray drift and used on any side of a field adjacent to a water body to protect aquatic organisms, or on all sides of a field to protect arthropods and non-target plants. If a 50 m buffer strip was required on one side of a field adjacent to a water body, there would be an approximately 8% loss in field area, which would be a loss of 16.8 t wheat per field (based on the average wheat field size in the UK (25 ha) and the average weight of wheat per ha (8.4 t)). If every wheat growing field in the UK required a 50 m buffer there could be a loss of ~16,000 ha of cropable area, which equates to ~134,000 t of wheat at a cost of more than £22 million (€29 million, $32 million). Due to a large proportion of its land area being underdrained, many arable farms in the UK have fields that are surrounded by open ditches (Droy, 2010), and therefore it is conceivable that many fields may require buffer strips on more than one side of a field. A 20 m buffer all around the field to protect terrestrial arthropods or non-target plants and related ecosystem services from spray drift, would result in approximately 16% loss of cropping area or 33.6 t of wheat per field, which at the UK scale is a loss of ~32,000 ha, ~238,800 t, ~£44 million (€59 million, $64 million).

We also found few alternative pesticides were available, should the buffer requirements associated with the most appropriate pesticide be unacceptable to the farmer or the regulatory authority. Indeed not all EU Member States, including the UK, currently allow these larger buffer strips. Alternative fungicides exist with narrower buffer strips, however, alternative insecticides for the study crops have buffer strip width requirements that are likely to be unacceptable to farmers or the regulatory authority in the UK (i.e. ≥20 m) (Table 2). In the UK 50% of the 2 million hectares of wheat grown is treated for black grass (Kleffmann Group, 2015). However, the viable alternative herbicides in wheat and in oilseed are not suitable to control black grass. In addition, propyzamide, the key herbicide used in oilseed for treating black grass in wheat, has been found exceeding acceptable levels in Drinking Water Protected Areas set by Article 7 of The Water Framework

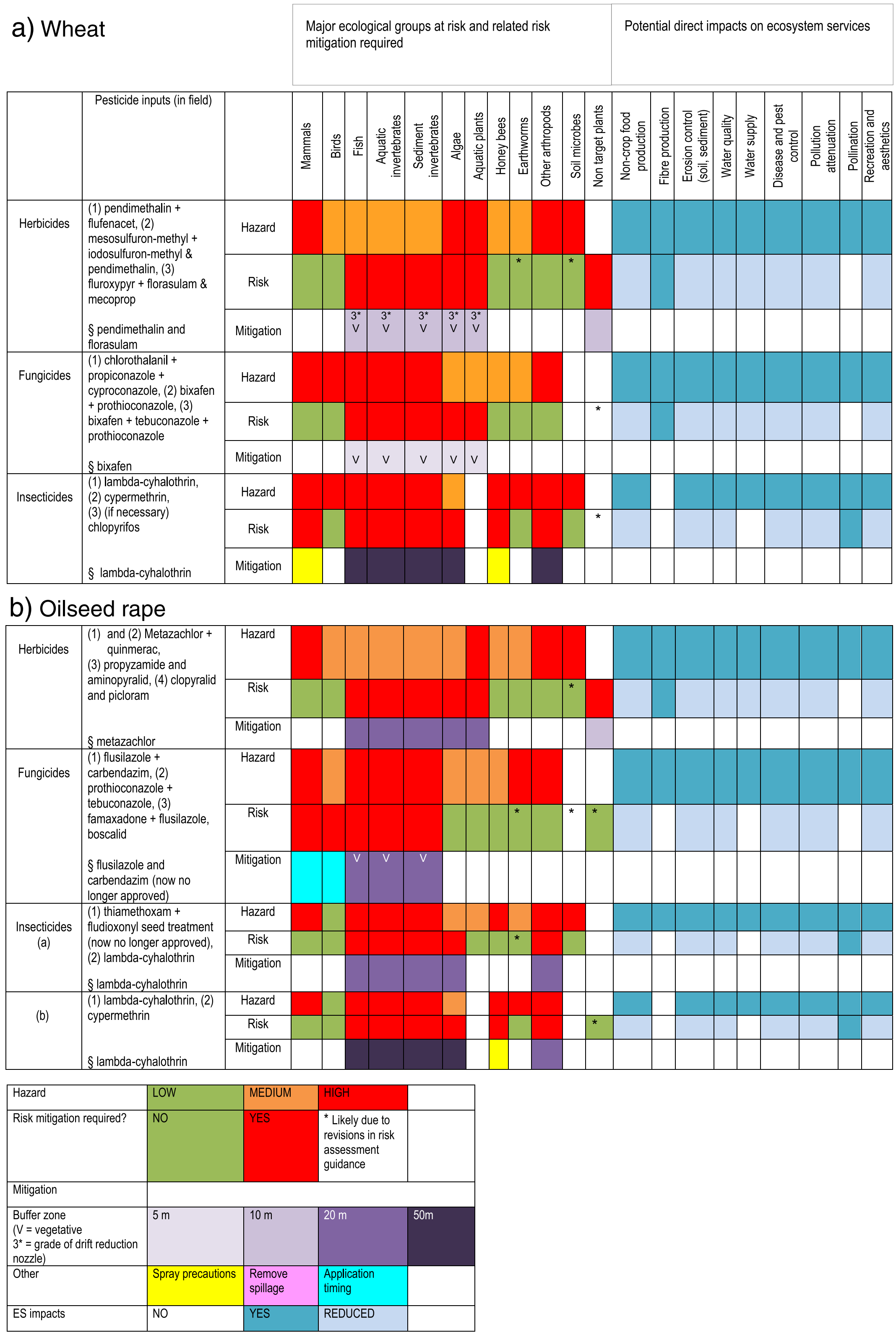

Fig. 2. The hazard, risk and mitigation measures currently recommended for the most commonly used herbicides, fungicides and insecticides used on (a) winter wheat (b) oil seed rape and (c) winter beans, with the likely consequent impacts on key arable ecosystem services. § indicates the substances that have driven the reported level of hazard and risk. The hazard level for non-target terrestrial plants could not be identified as this group has not been considered to be at risk until recently.

Directive (2000/60/EC). Farmers may shift away from any crop where the yield loss proves financially unacceptable, which could be the case for all three of the main crops investigated here. In particular, there may be potential difficulties growing oilseed rape since the moratorium of neonicotinoid seed treatments, as the peach-potato aphid, the vector of TYV, and the cabbage stem flea beetle can cause significant loss of yield (see above), the former being highly resistant to the alternative insecticides available.

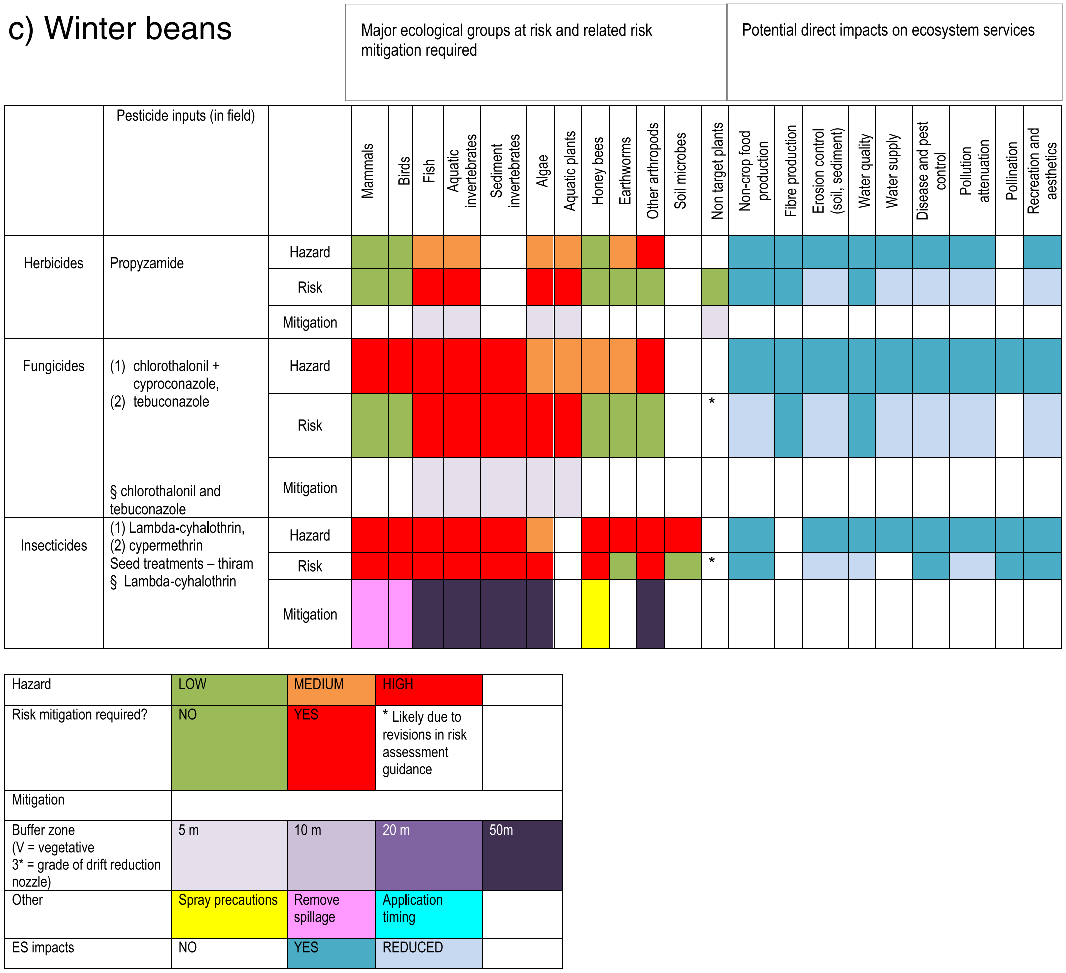

Fig. 2 (*continued*).

4. Discussion

In demonstrating the framework we have revealed a situation where pesticide policy designed to address conservative risk assessment schemes may now have far reaching and multiple consequences for farm businesses and food security in the UK. In the UK the value of wheat production was £2.1 billion in 2013 (DEFRA, 2015). Approximately 507,000 t of wheat per annum is consumed, 84% of which is home grown (DEFRA, 2015). However, with effective pesticides unavailable this percentage is likely to decline. With regards to the implementation of additional risk mitigation to ensure that the expected protection levels are met, all are not in use yet in the UK. For example, mitigation technology to reduce spray drift, which in turn enables farmers to reduce buffer width requirements, has not been authorised for use in the UK (they currently only accredit nozzles that deliver 75% reduction of spray drift, i.e. 3*, although 99% spray drift reduction nozzles are available and accredited in some other EU member states). Similarly, the consideration of vegetated buffer strips instead of bare soil to intercept spray drift is not yet considered in the risk assessment or as a risk mitigation option, in spite of their potential to further reduce pesticide transfer and increase biodiversity in the farmland due to their role in providing habitat and food resource (Hackett and Lawrence, 2014). Thus, a consequence of the simplified hypothesis underlying risk mitigation is that UK farmers may be subjected to wider buffer zones, and therefore greater reductions in crop yield, than farmers in other parts of Europe applying the same product to the same crop. All this could mean a potentially greater reliance on imported grain, in turn transferring the impact of production elsewhere, which may be greater in countries outside of the EU with more risk management options, or less stringent risk assessment.

Our analysis has highlighted the consequences of current risk assessment decisions on the ability to maintain crop yields whilst safeguarding a suite of important ecosystem services. When considering the implications of future changes in risk assessment, the challenge of maintaining food production whilst protecting ecosystem services and biodiversity will increase. For example, revisions suggested for the calculation of exposure levels (Predicted Exposure Concentrations) in soils (EFSA, 2015) and the definition of the endpoints to be used in a risk assessment of non-target terrestrial plants (EFSA, 2014) could result in risk assessment quotients one or two orders of magnitude lower than in the present assessment. Should the EFSA opinion (EFSA, 2015) recommendations for soils be accepted, we found that 12 of the most commonly used substances we have assessed (6 herbicides, 4 fungicides and 2 insecticides) will fail the risk assessment. This affects many of the herbicides used in wheat, oilseed and beans, again making black grass control difficult. It also affects fungicide use in wheat and beans, and insecticides use in wheat and oilseed. In addition to this, the main insecticide used in all the crops (Lambda-cyhalothrin) has now been renewed for a limited period of time (7 years) only. This continued narrowing of available pesticide chemistries and the consequences for which type of crop can be grown, may also have implications for meeting the requirements of the Common Agricultural Policy (CAP) in the UK. CAP reform may result in larger farms (>30 ha) being required to plant at least 3 crops with no more than 75% of the farmed area under one crop (DEFRA, 2014). Given our results this may be difficult to achieve, even meeting the two crop requirement may be

Table 2
Trade-offs between food production and ecosystem service provision within herbicide, fungicide and insecticide categories for each crop (winter wheat, oil seed rape and winter beans), including the issues with alternative chemistries and likely agronomic outcomes.

(i) Winter wheat	Trade-off
Herbicide	*Protect ecosystem services but suffer potential yield losses* Aquatic buffer will reduce crop area and is not reducible (spray drift reduction included in risk assessment). Terrestrial buffer will reduce crop area and is not reducible. Alternative herbicides not as effective against black grass.
Fungicide	*None* Buffer strip reducible using LERAP.
Insecticide	*Protect ecosystem services but suffer yield losses* Buffer will reduce crop area and is not reducible, alternative insecticides require 20 m buffer, switch crop if yield losses are high.
Issues with alternative chemistries	Herbicides not as effective against black grass. Insecticides require a 20 m aquatic buffer.
Outcome	*Loss of yield likely. Switch to another crop if loss proves unacceptable.*
(ii) Oil seed rape	*Trade-off*
Herbicide	*None* Buffers will reduce crop area and are not reducible. Alternative herbicides available but not as effective against black grass.
Fungicide	*None* Buffer not reducible but alternative fungicides available.
Insecticide	*Protect ecosystem services but suffer yield losses* Buffers will significantly reduce crop area and is not reducible. Alternatives not viable, a switch of crop likely.
Issues with alternative chemistries	Alternative insecticides to scenario (a) = (b). Alternative to (b) requires large buffer (aquatic 100 m) that is not reducible.
Outcome	*Loss of yield likely. Switch to another crop likely.*
(iii) Winter beans	*Trade-off*
Herbicide	*None* Buffer strip is reducible using LERAP. This herbicide is necessary for black grass control.
Fungicide	*None* Buffer strip is reducible using LERAP.
Insecticide	*Protect ecosystem services but suffer yield losses* Buffers will significantly reduce crop area and are not reducible. Alternatives require buffer strips that may not be reducible, switch of crop likely.
Issues with alternative chemistries	Insecticides require large buffers (100 m/20 m) for aquatics.
Outcome	*Loss of yield likely. Switch to another crop if loss proves unacceptable.*

hard for farms with arable land of 10–30 ha. This makes the possibility of creating mosaics of different habitats for promoting multifunctional landscapes, and hence biodiversity, unlikely. However, this is exactly what is required and it has the potential to be achieved by taking a landscape scale perspective when assessing and manging pesticide risks.

4.1. Risk assessment and management at the landscape scale

Resolving trade-offs in arable ecosystems relies on creating a more nuanced application of pesticide risk assessment. Buffer strip widths are derived from a conservative risk assessment that assumes a high vulnerability to risks everywhere, and maximises the likely spray drift exposure. Alternatively, risk evaluations could be more realistic and ground truthed so their applicability to particular landscapes could be assessed, and mitigation or restriction targeted where it is actually needed. Advances in technological mitigation measures for applying pesticides (and other inputs) could then be used to increase the opportunities for biodiversity and ecosystem services protection alongside food production.

Certain risk mitigation approaches, for example vegetated field margins or recovery areas, should be seen as opportunities to enhance, rather than simply safeguard ecosystem services and biodiversity. A recent study of the EU CAP policy to ensure that 3–5% (up to 7% in 2017) of EU farmland is managed as an ecological focus area (fallow, buffer strips etc), found that landscapes with 3–7% of their area as natural elements had 37–75% of maximum species richness (Cormont et al., 2016). The balance of delivery between food production, other ecosystem services and biodiversity that can be achieved is likely to differ from one arable system to another depending very much on the characteristics of the surrounding land use matrix, as well as the social and economic contexts. A patterning of both intensive and low-impact farming areas (rather than ‘land sparing’ or ‘land sharing’ *sensu* Green et al., 2005) within and across landscapes is desirable to promote such multifunctionality. To ensure domestic food security can be attained, arable areas within landscapes, or indeed whole landscapes, may be best optimised for crop production, with less emphasis on providing other services. Such intensification of agriculture could, where possible, be confined to landscapes where impacts can be limited (i.e. areas that may be more robust to pesticide use and fertilizers, with fewer sensitivities or risks than in other areas), for example, in catchments that are not Drinking Water Safeguard Zones, or are of lower biodiversity conservation value (Shackleford et al., 2015). In contrast more multifunctional agricultural landscapes can be achieved where there are environmental sensitivities or nature protection areas, where acceptable risk levels should be much lower, and/or where payments for ecosystem services (PES) schemes can be identified to incentivise particular farmer behaviours to enhance a suite of services (Russi et al., 2016). However, this requires mapping and understanding how the spatial location, or patterning, of arable land use and green infrastructure across farms and landscapes might influence the number and quality of ecosystem services and enhance biodiversity (Liquete et al., 2015).

The implementation of this vision at this scale requires integration of land use decisions across farms to achieve landscape scale benefits, whilst ensuring that national level strategies can be met. Potential trade-offs would need to be identified with current knowledge when forming all environmental policies that directly or indirectly affect how land is managed. Indeed, taking such a perspective will be necessary for an effective implementation of the new EU Green Infrastructure Strategy (COM/2013/0249), that will eventually be filtered through all environmental policies. Regulations need to be spatially and temporally flexible to make sure we can optimise food production where it is most appropriate. For example, not necessarily enforcing blanket bans of substances if a water quality problem occurs that is local, short term and out of sensitive areas. A range of regulations, payments and charges appropriate to particular contexts will be required, that incentivise co-ordinated management across farms (Van Zanten et al., 2014). This may create networks that are more likely to take up Payment for Ecosystem Services schemes and changes to management to acheive landscape scale benefits (Van der Horst, 2011). In the UK, policy frameworks exist through which such an approach could be enabled, for example, the Water Framework Directive at the basin scale, Catchment Sensitive Farming, and the new Countryside Stewardship Scheme (which to some extent should encourage co-operation across neighbouring farms). Taking this framework forward and using it in a landscape context will help in balancing environmental protection goals with socio-economic considerations at the landscape level, the responsibility of European Commission and Member State risk managers. Making progress here will improve the much needed transparency in decision-making.

5. Conclusions

Revealing the trade-offs between food production and ecosystem services when using pesticides allows the implications of policy change and risk assessment to be evaluated. We show that the current changes in EU legislation, risk assessment and the limitations associated with mitigation measures may have serious implications for food production, and management to enhance ecosystem services and biodiversity. It is important to be aware of the consequences of changes in agricultural policy, and in many cases this will mean accepting difficult trade-offs. The current restriction on the use of three neonicitinoid insecticides in the EU is a case in point. The moratorium has been made to protect insect pollinators and the pollination service they provide, but it has potential consequences for the viability of oilseed as a break crop, and therefore on the ability to control black grass in wheat production. Pollination is an important ecosystem service and wheat is a major staple food. How to balance the need for food security and the protection of ecosystem services such as pollination should be a key consideration when managing pesticide risks. Clearly we can't have it all everywhere. Therefore, identifying options and formulating policy to create incentives for mitigation measures that enhance arable system multifunctionality where possible within and across landscapes, is the only realistic way 'to have it all' at least in some places.

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Appendix A. Supplementary data

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