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Perspective

Optimizing agri-environment schemes for biodiversity, ecosystem services or both?

Johan Ekroos [a,*], Ola Olsson [b], Maj Rundlöf [b,c], Frank Wätzold [d], Henrik G. Smith [a,b]

[a] Centre for Environmental and Climate Research, Lund University, Ecology Building, 223 62 Lund, Sweden
[b] Department of Biology, Lund University, Ecology Building, 223 62 Lund, Sweden
[c] Department of Ecology, Swedish University of Agricultural Sciences, Box 7044, 750 07 Uppsala, Sweden
[d] Chair of Environmental Economics, Brandenburg University of Technology Cottbus-Senftenberg, Postbox 101344, 03013 Cottbus, Germany

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ABSTRACT

Agri-environment schemes (AES) have been introduced to mitigate negative environmental effects caused by increased agricultural intensification in Europe. However, there is still debate on whether currently available incentives are efficiently enhancing farmland biodiversity. Moreover, agri-environment schemes often lead to a yield reduction, which has been argued to potentially increase pressure on non-cropped habitats, with unintended negative environmental consequences. Here, we argue that AES should build on more explicit goals regarding (1) biodiversity protection as such and (2) provisioning of ecosystem services benefiting agricultural production. We discuss how this can be achieved by an efficient spatial allocation of AES measures to the benefit of biodiversity, ecosystem service providers and agricultural production. We differentiate between biodiversity conservation schemes, which target species of conservation concern, and ecosystem service schemes which explicitly target ecosystem service providers important for environmentally sustainable agriculture, most of which are common species. We construct a simplistic, conceptual model, based on well-founded ecological principles, to illustrate how to allocate biodiversity conservation schemes and ecosystem service schemes spatially, depending on where they are needed in order to meet the goals of protecting biodiversity per se and promoting environmentally sustainable agriculture. By understanding the functional importance of different types of AES we can achieve much more effective schemes in the future.

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1. Introduction

Agricultural production is expected to increase tremendously during the next 35 years driven by increased demands for food (Godfray et al., 2010; Tilman et al., 2011) and biofuel (Miyake et al., 2012). Although efficient reductions of food waste may reduce demands for increased production to some extent (Tscharntke et al., 2012a), this is a worrying prospect, since agricultural expansion and intensification are major drivers of biodiversity loss (Green et al., 2005; Stoate et al., 2009), biotic homogenization (Ekroos et al., 2010; Karp et al., 2012) and changes

* Corresponding author. Tel.: +46 462228630.
E-mail addresses: johan.ekroos@cec.lu.se, jeekroos@gmail.com (J. Ekroos).

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in community composition and functional diversity (Bommarco et al., 2012; Flynn et al., 2009). This does not only compromise conservation goals, but may also ultimately undermine the provisioning of many ecosystem services which depend on biodiversity (Cardinale et al., 2012). It is therefore critical that this biodiversity loss is mitigated in agricultural landscapes (Foley et al., 2011; Godfray et al., 2010).

In Europe, agri-environment schemes (AES) are implemented to reduce and counteract further declines in farmland biodiversity (Anonymous, 2005). In many countries AES are not targeted at particular species of conservation concern and might not provide a general solution for biodiversity conservation (Kleijn et al., 2011). AES may also negatively affect yields, e.g. through land-use opportunity costs (e.g. conservation headlands, Kaphengst et al., 2011), restrictions on agricultural management (e.g. mowing date Kaphengst et al., 2011) or reductions in farming intensity (e.g. organic farming, Gabriel et al., 2013; Seufert et al., 2012), and low yields at existing farmland has in turn been argued to further increase pressure for converting land into arable production elsewhere due to leakage effects (Balmford et al., 2012; Phalan et al., 2011; Tilman et al., 2011). As a consequence, it has been argued that many widely implemented AES can create unintended negative consequences for biodiversity (Balmford et al., 2012; Phalan et al., 2011).

How can we find a balance between these multiple pressures and demands and the pressing need for environmentally sustainable agriculture? Here, we argue that AES, which provides the policy framework for sustainable agriculture in Europe, should build on more explicit goals regarding (1) biodiversity protection as such and (2) provisioning of ecosystem services for environmentally sustainable agriculture (ecological intensification sensu Bommarco et al., 2013). Secondly, we argue that AES with these different targets should be spatially implemented with reference to each other, accounting for emergent trade-offs and synergies between them (Macfadyen et al., 2012).We conceptually identify the optimal spatial allocation of targeted AES to maximize conservation benefits while maintaining a high level of agricultural production (sensu Bommarco et al., 2013). We use this conceptual framework to investigate how the different AES should be implemented in landscapes differing in productivity, which is a variation that has been considered important for the uptake of AES (Gabriel et al., 2009; Rundlöf and Smith, 2006).

2. A typology of agri-environment schemes

A general principle behind AES is to decrease management intensity (the timing, frequency and intensity of mechanical disturbance and application of pesticides and mineral fertilizers) by managing fields or parts of fields less intensively (e.g. by introducing buffer strips, managing grasslands with low agrochemical inputs, or implementing organic farming) or managing existing non-crop habitats according to certain prescriptions (e.g. hedgerow management, management of pastures or semi-natural grasslands). Two general hypotheses on where to implement AES has received wide attention in the literature (Kleijn et al., 2011; but see Carvell et al., 2011; Gabriel et al., 2013). The first hypothesis suggests that the effectiveness of AES to promote biodiversity is high in landscapes with intermediate landscape complexity (Concepción et al., 2012; Rundlöf and Smith, 2006; Tscharntke et al., 2005, 2012b) because the effectiveness of AES is low both in landscapes deprived of source habitats for organisms to colonize the scheme area, and in complex landscapes with high availability of source habitats where biodiversity will be high irrespective of local management. In contrast, another hypothesis suggests that a decelerating loss of biodiversity with increased agricultural intensity makes AES most effective in areas where biodiversity is high, since the marginal gain from a reduction in intensity is largest in these landscapes (Gabriel et al., 2010; Kleijn et al., 2009; Kleijn and Sutherland, 2003; Whittingham, 2011).

We suggest that the apparent disagreement between the two hypotheses presented above largely depends on what part of overall biodiversity is targeted (Kleijn et al., 2011; Rey Benayas and Bullock, 2012; Smith et al., 2010). Some AES are explicitly targeted for protecting biodiversity. Such schemes are typically focused on habitat protection for species of conservation concern, and we refer to AES of this type as *biodiversity conservation schemes*. Apart from some species originally adapted to various steppe habitats, most notably some farmland birds (Wright et al., 2012), the majority of species of conservation concern cannot establish viable populations outside their main habitat (Hanski and Ovaskainen, 2000), such as in cultivated farmland (Balmford et al., 2012; Phalan et al., 2012). A prominent example of a biodiversity conservation scheme is support to manage semi-natural grasslands, with the explicit intention to maintain high biodiversity values (Franzén and Nilsson, 2008; Hodgson et al., 2010). Managing these grasslands is highly important for conservation since many grassland species are strict habitat specialists (Ekroos and Kuussaari, 2012). For example, in Sweden, semi-natural grasslands have been classified according to whether they have specific biodiversity values (Jordbruksverket, 2005), and those classified as having such values are eligible for a higher monetary compensation for management efforts. Historically, semi-natural grasslands have experienced major losses in Europe through conversion to arable farming, intensively managed, cultivated grasslands, or forest (Bullock et al., 2011; Poschlod and WallisDeVries, 2002). Increasing demand for food and biofuels increases incentives for farmers to convert semi-natural grasslands (Bowyer, 2010; Miyake et al., 2012).

In contrast, many AES are not explicitly targeted at protecting specific species of conservation concern but focus on more general goals related to other environmental benefits and ecosystem services, e.g. to improve water quality. Managing the provisioning of ecosystem services such as pollination and biological pest control may contribute to the ecological intensification of agriculture, where regulating and supporting ecosystem services replace environmentally harmful anthropogenic inputs (Bommarco et al., 2013). Many ecosystem service providers are common habitat generalists which are not as restricted by the availability of particular habitats, and therefore occur in a wide variety of environments (Gaston and Fuller, 2008; Kremen et al., 2007). We refer to AES explicitly targeting ecosystem services as *ecosystem service schemes*. Specific examples of ecosystem service schemes are beetle banks and flower strips, which are often specifically intended to benefit mobile ecosystem services providers (Griffiths et al., 2008; Scheper et al., 2013).

Many existing AES are not easily classified as biodiversity conservation or ecosystem service schemes, such as those targeted at particular bird species (Whittingham, 2011). They may be very successful in terms of bird conservation (Perkins et al., 2011; Baker et al., 2012), but they may also benefit ecosystem service providers (Wilkinson et al., 2012). Indeed, we can assume that biodiversity conservation schemes benefit both species of conservation concern and ecosystem service providers, but that ecosystem service schemes mainly benefit the latter group (Kleijn et al., 2011; Macfadyen et al., 2012; Tscharntke et al., 2005; but see Pywell et al., 2012). It is also widely assumed that the most dominant species in a given community has a disproportionately large effect on ecosystem functions (Díaz et al., 2007; Geider et al., 2001; Grime, 1998), although rare species may be functionally important in some cases (Mouillot et al., 2013; Walker et al., 1999).

Importantly, the separate goals of biodiversity conservation versus ecosystem service provisioning dictate at which spatial