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Viewpoint

Why do we need to integrate farmer decision making and wildlife models for policy evaluation?

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ABSTRACT

Environmental and agricultural policy instruments cause changes in land-use which in turn affect habitat quality and availability for a range of species. These policies often have wildlife or biodiversity goals, but in many cases they are ineffective. The low effectiveness and the emergence of unwanted side effects of environmental and agricultural policies are caused by over-simplistic assumptions in the design of policy instruments as well as difficulties with predicting behaviours of policy subjects. When considering wildlife in agricultural landscapes, policy's performance depends both on human (farmers) actions, which the policies aim to affect, and wildlife responses to land-use and management changes imposed by farmers. Thus, in order to design effective agri-environmental policies, detailed *ex-ante* assessments of both of these aspects are necessary. Due to the restrictive assumptions and technical limitations, traditional agricultural economic and ecological models fall short in terms of predictions of impacts of agri-environmental measures. The feedback situation between policy, human behaviour and ecological systems behaviour can confound these approaches, which do not take systems complexity into account. Therefore, a solution that integrates both feedback interactions and the differing scales at which these interactions take place is needed. For this, we suggest developing integrated policy assessment tools comprising of simulated farmer decision making, on-farm land-use and wildlife responses in the form of spatially explicit, dynamically connected agent-based models. Although complex and necessitating true inter-disciplinarity, these approaches have matured to the point where this endeavour is now feasible.

Introduction

The importance of sustaining and enhancing biodiversity has been recently underlined in many international initiatives (e.g. 2011–2020 – the United Nations Decade on Biodiversity (Anon, 2011c), as a follow up on the Strategic Plan for Biodiversity 2011–2020 (Anon, 2010b), European Parliament Resolution on the EU 2020 Biodiversity Strategy (Anon, 2012) and EU biodiversity strategy to 2020 (Anon, 2011b)). In the context of the EU agricultural policy, biodiversity has been present in the EU regulations since the introduction of optional agri-environmental measures in 1985. In 1992, it became obligatory for all Member States to introduce such schemes. In 2010, the need for better biodiversity management was stated as one of the reasons for introducing the upcoming CAP (Common Agricultural Policy) reforms (the CAP towards 2020, (Anon, 2010a)). The problem of 'enhancing farmland habitats and Biodiversity' is and will be an area of major concern according to the impact assessment of the recent CAP 2020 reform proposals (Anon, 2011a). Despite the fact that there have been several measures aimed at environmental protection within CAP, e.g. decoupling, cross-compliance and agri-environment schemes, the loss of farmland biodiversity continues (Anon, 2012).

Biodiversity related policies have not succeeded in achieving their goals; this has also often happened with other environmental policies. An example from Denmark illustrates problems of agricultural policies aimed at changing farmers' behaviour. In 1996, the tax rate for pesticides was significantly increased to achieve a reduction in pesticide usage of 50% in 1997 compared to the period 1981–1985 (Anon, 2001). The pesticide treatment index (a metric used to measure relative pesticide usage) in 1997 remained on the same level as in 1981–1985 (Anon, 2001). In 1998, a subsequent raise in the tax rate achieved only around 10% reduction compared to the reference period. Obviously, farmers' response to the tax was not as strong as expected by policy makers. Additionally, introduction of the tax had 'secondary' effects, e.g. change in pesticide application patterns or changes in crop composition. These changes were shown to have indirect negative impacts on a population of skylarks (Jacobsen et al., 2008).

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The example points to two problems related to environmental policies: failure in achieving intended effects and emergence of unintended side effects. Thus, it is important also to pay attention to unintended affects not least in the case of policies targeted at agricultural systems, as the unintended damaging effects on environment, e.g. loss of species, increase of ground water pollution may occur. In terms of biodiversity policies, Maestre Andrés et al. (2012) refer to these effects as rebound effects which decrease the conservation benefits, thus causing policies to be less effective. Due to the lack of proper *ex-ante* impact analysis (or its absence) there have been policies that provoked unexpected outcomes, both in terms of human behaviour and environmental impacts. Examples of unintended, negative side effects of environmental or agricultural policies vary widely and include e.g. relocation of pollutants (Raadschelders et al., 2003) or pollutant emitting activities (Kinzig et al., 2011), net increase in CO_2 release as a result of land conversion for purpose of biofuel production (Fargione et al., 2008), abatement of one chemical at an expense of environmental effects of another (Raadschelders et al., 2003), illegal harvest (Cinti et al., 2010), increased pollution from agriculture (Dent et al., 1995).

In order to ascertain whether a particular policy instrument can achieve its goals, has any unexpected 'side effects' or even adverse outcomes, a proper impact assessment is necessary. To achieve the type of impact assessment called for by the EU Commission (Anon, 2002) for biodiversity policies (i.e. an integrated method for assessing economic, social and environmental effects of proposed policy regulations), we require linkage of multiple aspects of the problem. These include: how farmers, as subjects of agricultural policies, behave in response to them; how this translates into farm management and thus changes in agricultural landscape; and finally, what are the impacts of the changes in landscape on biodiversity. Direct integration of these aspects could improve systems understanding, directly warn policymakers of potential failures and show the reasons behind them, as well as assist in designing effective measures of promoting biodiversity.

In this article we argue that there is a need for developing integrated models for assessing alternative policy instruments for their impacts on wildlife. We discuss the specific conditions these models have to fulfil in order to provide reliable results. The focus is on the development of decision making models based on realistic assumptions and on the importance and technical problems arising from the integration of models of different system components.

The first section deals with features of environmental policy instruments which might reduce their effectiveness and hinder policy impact assessment. Next, we show the characteristics of existing agricultural models that are responsible for these models' low accuracy at small scales and thus, reduce their usefulness for integrated agricultural–ecological assessments. The subsequent section reviews on the impacts of changes in land-use on wildlife. Finally, we focus on specific problems and recommendations for integrated social–ecological models.

Problem definition

The problem of agricultural policies and their impacts on wildlife effectively consists of a chain of interacting components, i.e. a policy has an impact on farmer's decisions; the farmer alters management which results in landscape changes; and landscape changes have impacts on wildlife. To properly assess agricultural policies with respect to their impacts on wildlife it is therefore crucial to properly represent (model) each of these levels and linkages between them. If the assessment fails at any of the levels, the goals of policies might not be achieved.

Problems with assessing the impacts of environmental policy instruments

Assumptions behind environmental policy instruments

The major problem with assessing environmental policy instruments is associated with the assumptions behind them.

Environmental policies achieve their goals by affecting the behaviour of entities or individuals, e.g. power plants emitting pollutants or farmers. Various types of instruments might be used for this purpose. Command and control (CAC) approach instruments enforce particular behaviours (e.g. by imposing emission limits, setting maximum allowed fertiliser usage or banning use of certain pesticides). Alternatively, market-based instruments (MBIs) aim at giving economic incentives (e.g. tax reductions or subsidies on environmentally friendly technologies) to steer decisions of targeted subjects in a desired direction.

The assumptions behind MBIs are that policy subjects are profit oriented utility maximisers (Schneider and Ingram, 1990; Mikael Skou Andersen and Sprenger, 2000), who are able to make rational choices, i.e. have enough information and skills to select alternatives that maximise their utility. This implies that subjects choose actions that maximise their profit. CAC instruments assume or appeal to different types of motivation, i.e. moral obligation (duty, non-utilitarian), economic utility (cost–benefit calculations) and social norm following. The economic utility perspective assumes that subjects assess the utility they get from complying with a regulation (e.g. lost income or extra costs) as well as the consequences of not complying (e.g. a fine multiplied by a probability of being charged); choosing the action characterised by a higher utility. Following social rules could also be incorporated into utility calculations (see e.g. Lindbeck (1997) on negative utility from social norm deviation), although these need not necessarily be monetised.

Making the assumption that policy subjects are fully rational may lead to erroneous conclusions about the outcomes of both types of policy instruments (MBIs and utilitarian-based CAC). There exists both a theoretical basis, i.e. bounded rationality theory (Simon, 1955) and empirical evidence for people not necessarily being fully rational (e.g. Nielsen, 2009). If policy subjects are not rational, i.e. do not calculate their utilities correctly, in the case of CACs the rate of non-compliance might be higher than expected by policy designers or, in the case of MBI, the strength of the financial incentive might be not enough to induce desired behaviours. For MBIs instruments, their effectiveness might also be impaired by the fact that subjects consider other than strictly economic factors in their decision making (Nielsen, 2012). If the proportion of subjects not complying with CAC or not reacting to MBI is significant, the policy goals will not be reached. Thus, it is important to assess *ex ante*, what behaviours can be expected in response to introduction of environmental policy (Dent et al., 1995), whether it introduces an MBI or CAC instrument.

Problems with predicting policy subjects' behaviours

Another problem of policy impact assessment is that policy makers often do not consider the whole spectrum of behaviours that may emerge as a result of introducing a new policy instrument. The unexpected behaviours might work both in favour and against the policy goals as well as affect areas not targeted by the policy. For instance, initially set-aside schemes in the EU were introduced solely to reduce agricultural production; as they proved to have beneficial effects on biodiversity, they were subsequently also used as means of enhancing biodiversity. These behaviours might be 'creative' solutions that cannot be easily discovered by control authorities. Hypothetically though, the unexpected or rather not considered behaviours might simply be a consequence of the behaviours that were desired by the policy. For example, a farmer