



Short Communication

Concave trade-off curves between crop production and taxonomic, functional and phylogenetic diversity of birds

Camille Dross^{a,b,*}, Frédéric Jiguet^c, Muriel Tichit^a^a UMR SADAPT, INRA, AgroParisTech, Université Paris-Saclay, 75005 Paris, France^b AgroParisTech, 19 avenue du Maine, 75015 Paris, France^c Centre d'Ecologie et des Sciences de la Conservation (CESCO UMR7204), Sorbonne Universités, MNHN, CNRS, UPMC, CP135, 43 rue Buffon, 75005, Paris, France

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ABSTRACT

In a context of land scarcity, food production and biodiversity conservation objectives compete for land. The shape of the relationship between these two objectives may be helpful to inform decision-making. However, the metrics used to evaluate this relationship have so far been restricted to species abundances and species richness, which give no information on possible consequences on ecosystem functioning or on evolution history. Indeed, the shapes of the relationship between food production and other diversity facets, such as functional diversity and phylogenetic diversity, have rarely been studied. We considered 3 diversity facets: taxonomic diversity, functional diversity and phylogenetic diversity. For each facet, several biodiversity metrics have been proposed. The objective of this work was to investigate whether the shape of the trade-off curve between food production and biodiversity metrics depended on the considered facet of biodiversity. Using data from the national agricultural statistics, we computed edible energy from crops and from livestock on a nation-wide gradient covering French agroecosystems. Using bird observation data provided by the French Breeding Bird Survey (FBBS), we computed 9 (3 for each facet) biodiversity metrics in 516 different sites of the FBBS. The trade-off curves were then computed using additive mixed models. All metrics decreased along a crop production gradient. For functional and taxonomic diversity metrics, the slope was steeper at high levels of production, suggesting that actions aiming at increasing local taxonomic or functional diversity may be more efficient in regions with highest crop production, if restoration follows the same trajectory as biodiversity loss. The decrease was steeper for functional diversity than for taxonomic diversity, suggesting that agriculture can reduce the functional diversity of birds more than their taxonomic diversity.

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1. Introduction

Agriculture is a major driver of biodiversity loss, both through intensification and through agricultural expansion. Actions aiming at halting or reversing this loss will need to be carefully targeted to be as cost-effective as possible (Batáry et al., 2015). A body of literature indicates that actions aiming at restoring local diversity are most effective in landscapes with little or with intermediate complexity (Tscharrntke et al., 2005; Concepción et al., 2012; Tscharrntke et al., 2012; Cormont et al., 2016). Conversely, the importance of conserving existing diversity in so-far preserved or extensive land-

scapes has often been stressed (eg. Phalan et al., 2011; Kampmann et al., 2012). Drawing insight from evolutionary theory, Charpentier (2015) argues that the shape of the trade-off curve, i.e. the relationship between production and biodiversity, may be important to define optimal solutions.

Biodiversity or biological diversity is the variability among all living organisms. It includes within-species diversity, such as genetic diversity, diversity between species, and variability of ecosystems (United Nations, 1992). In the case of production/biodiversity trade-off curves, it is easiest to focus on diversity between species. Most of the articles studying productivity/biodiversity trade-off curves use the number of species present (hereafter species richness) or some species' abundances as biodiversity metrics (Kremen, 2015; Law and Wilson, 2015).

Beyond species richness, numerous ways to measure biodiversity have been proposed (Pavoine and Bonsall 2011), making it possible to consider different facets of biodiversity. Following

* Corresponding author at: UMR SADAPT, INRA, AgroParisTech, Université Paris-Saclay, 75005, Paris, France.

E-mail addresses: camille.dross.2008@polytechnique.org (C. Dross), muriel.tichit@agroparistech.fr (F. Jiguet), fjiguet@mnhn.fr (M. Tichit).

Devictor et al. (2010), we will henceforth use the term “diversity facets” to refer to taxonomic diversity, functional diversity, or phylogenetic diversity. The shape of the relationship between functional diversity or phylogenetic diversity and agricultural production has not been assessed, even though both functional diversity and phylogenetic diversity are perceived as particularly important. Indeed, species’ functional characteristics drive ecosystem functioning and influence ecosystem service supply at present (Hooper et al., 2005). In the future, an ecosystem’s capacity to cope with changes and sustain functions in the long term may depend on the evolutionary potential of its species (Sarrazin and Lecomte, 2016). Community assembly schemes such as environmental filtering or competition may cause functional diversity to diverge from taxonomic diversity (Mouchet et al., 2010). Flynn et al. (2009) report that the loss of functional diversity associated with agriculture can be more severe than it would be if it simply reflected the loss of species richness. They suggest that this result might be due to environmental filtering where only species with a specific set of traits can survive in agricultural landscapes.

For each facet, numerous metrics have been proposed (Mouchet et al., 2010; Winter et al., 2013). Some indices are redundant, others carry different information, such as richness and evenness (Schweiger et al., 2008; Mouchet et al., 2010). Some integrate information about species’ relative abundances while others merely reflect the range of items (traits, phylogenetic branches) present. Although such diversity indices are often driven by species richness (Poos et al., 2009), the relationship between taxonomic diversity and phylogenetic or functional diversity is variable (Calba et al., 2014). A unifying framework based on Hill numbers has recently been proposed (Chao et al., 2014). Hill numbers are the effective numbers of taxonomic entities (species), phylogenetic entities (branches of unit-length), or functional entities (species-pairs with unit-distance between species), and their advantage is that they provide a common framework for all 3 diversity facets. Hill numbers of order 0 do not take into account species abundances, but only presence-absence. Hill numbers of order 1 take abundances into account, and Hill numbers of order 2 are based on squared relative abundances. Numerous indices previously proposed can be transformed into one of these indices.

The objective of this work was to investigate whether the shape or the strength of the trade-off curve between food production and biodiversity metrics depended on the considered diversity facet. We considered two hypotheses:

H1. Functional diversity declines more steeply along a production gradient than taxonomic diversity, possibly because of environmental filtering (Flynn et al., 2009) or because of a higher competition at low production levels.

H2. Phylogenetic diversity declines less steeply along a production gradient than taxonomic diversity, possibly because the traits that make species vulnerable to agriculture are phylogenetically dispersed (Mouquet et al., 2012).

To test these hypotheses, we computed agricultural production and 9 biodiversity metrics across French agroecosystems and we used generalized additive mixed models to assess the relationship between each biodiversity metric and production. Simple linear models with functional or phylogenetic diversity as the dependent variable and production and taxonomic diversity as the predictor variables then made it possible to assess differences in the strength of the responses to production.

2. Materials & methods

2.1. Agricultural data

Edible energy from crops and from grazing livestock (cattle, goats, and sheep) were computed for 244 Small Agricultural Regions (SARs) that had a mean area of 1 418 km² and for which bird data were available in agricultural habitat. French SARs are consistent with administrative boundaries and have homogeneous soil-climatic conditions. Data on areas under cultivation for each crop type and on animal numbers were obtained from the 2010 Agricultural Census, which is public data except when confidentiality rules are applicable, in which case estimations were carried out to recover the information missing due to confidentiality requirements. Information about the estimation process can be found in Appendix A. Data on crop and animal production were derived from 2010 annual statistics. This data set was available at the Nomenclature of Territorial Units for Statistics (NUTS) 3 level (mean area = circa 5800 km²), and was disaggregated using data on areas under cultivation for each crop type and on animal numbers. Crop yield and animal productivity were assumed to be constant within each NUTS 3 level. Production data were converted into energy value using conversion coefficients (FAO, 2003; ANSES, 2013) and divided by the agricultural area in the SAR to account for different SAR areas.

2.2. Biodiversity data

We focused on common birds because they are generally sensitive to change (Jiguet et al., 2007) and because their characteristics may reflect changes occurring in taxa they feed on (Wilson et al., 1999). Additionally, they are widespread and commonly surveyed (Jiguet et al., 2012), they provide numerous services (Wenny et al., 2011; Sekercioglu 2012) and they have been suggested as an accurate gauge for measuring environmental health (Gregory and van Strien, 2010). We focused on 106 common breeding bird species listed in Appendix B.

2.2.1. Abundance data

All bird data were taken from the French Breeding Bird Survey (FBBS). The FBBS is a standardized monitoring program in which skilled volunteer ornithologists count breeding birds in randomly selected sites across the country (Jiguet et al., 2012). Each FBBS observer is randomly allocated a 2 × 2 km site, in which he or she positions 10 point counts. Twice every spring (before and after May the 8th, at least 4 weeks apart), the observer visits the site and performs a five-minute survey at each point count. Every individual bird either heard or seen is recorded along with the distance of contact (<25 m, 25–100 m, >100 m).

Databases describing land use within each FBBS site (provided by Sausse et al., 2015) made it possible to select within the 244 SARs 516 FBBS sites that had a least half their area in agricultural lands. As agricultural activities are relatively slow changing and to ensure our results were not dependent on conditions during a particular year, we used bird data from years 2010 to 2013. For each year and each site, we computed abundances for each species in 3 steps. First, we retained the maximum abundance over the two yearly surveys for each point count, except for three species which generally have not completed their spring migration at the time of the first yearly visit. For these species we considered only counts from the second visit (Princé et al., 2015). Second, for less than 0.1% of point counts, more than ten individuals of a single species were recorded during a single five-minute point count. As it is difficult for observers to count accurately large numbers of similar individuals, we only took into account the ten first individuals of each species observed during a single point count. Third, we summed abundances over all

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