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Short communication

Risk of pesticide exposure for reptile species in the European Union☆

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ABSTRACT

Environmental pollution has an especially high impact on wildlife. This is especially the case in industrialized countries. Although, many species within the European Union benefit from protection by the Habitats Directive, no special consideration is given to possible detrimental effects of pesticides. This is in particular remarkable as negative effects, which may lead to a regional diversity loss, have already been identified in laboratory and mesocosm studies. We conducted a pesticide exposure risk evaluation for all European reptile species with sufficient literature data on the considered biological and ecological aspects and occurrence data within agricultural areas with regular pesticide applications (102 out of 141). By using three evaluation factors – (i) pesticide exposure, (ii) physiology and (iii) life history – a taxon-specific pesticide exposure risk factor (ERF) was created. The results suggest that about half of all evaluated species, and thus at least 1/3 of all European species exhibited a high exposure risk. At the same time, two of them (*Mauremys leprosa* and *Testudo graeca*) are globally classified as threatened with extinction in the IUCN Red List of Threatened Species. Variation regarding species occurrence in exposed landscapes between pesticide admission zones within the EU is rather large. This variation is mainly caused by differing land use and species abundances between zones. At the taxonomic level, significant differences in exposure risk can be observed between threatened and non-threatened species, which can be explained by the formers remote distribution areas. Lizards display the highest sensitivity toward pesticides, although no differences in overall ERFs can be observed between taxonomic groups.

By identifying species at above-average risk to pesticide exposure, species-based risk evaluations can improve conservation actions for reptiles from cultivated landscapes.

1. Introduction

Biodiversity decline is a global problem among all taxa and ecosystems (Stuart et al., 2010). In reptiles, as one of the various threatened vertebrate groups, ongoing worldwide population declines are recognized (Amaral et al., 2012a; Gibbons et al., 2000; Todd et al., 2010). The causes for these declines are highly assorted, and it is believed that among all factors, habitat loss and degradation is the major factor in industrialized countries (Gibbons et al., 2000; Todd et al., 2010), followed by agrochemical use in their habitats (Bicho et al., 2013; Gibbons et al., 2000; Todd et al., 2010; Weir et al., 2010). In the European Union (EU), 18% of all reptile species that have been evaluated by the IUCN Red List of Threatened Species in 2015 are considered as threatened, i.e. in the category "Vulnerable" or higher (Cox and Temple, 2009; IUCN, 2015).

Although effects of pesticides on reptiles have been reviewed to some degree, and different studies have shown evidence of potential strong effects on reptile wildlife (Amaral et al., 2012a,b; Cardone, 2015; Carpenter et al., 2016; Douros et al., 2015; Latorre et al., 2013; Poletta et al., 2016; Schaumburg et al., 2015; Weir et al., 2010, 2014, 2015; Willemsen and Hailey, 2001), there is still a great lack of data. Especially, toxicity data concerning squamates is scarce (Sparling et al., 2010; Weir et al., 2010), and data on effects of pesticides in species' natural habitats even more so. Additionally, reptiles are currently not considered for risk evaluation processes during pesticide admission procedures in the EU (EFSA, 2009; Regulation (EC) No 1107/2009). While knowing the potential effects of pesticides on reptile populations is indeed of great importance, knowing which reptile species and populations may come into contact with pesticides in the first place seems of equal significance. This approach would allow us to identify which species will suffer the most due to pesticide use, and could thus provide key data for conservation practice.

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So far, there have been few studies concerning herpetological biodiversity patterns within different croplands (Balouch et al., 2016; Carpio et al., 2016), but none of them actually investigated the potential exposure risk of reptiles towards pesticides, and they were either on a small scale, or did not target European reptiles, specifically. There has only been one study concerning the potential pesticide exposure risk of European reptile species by Wagner et al. (2015). In this study, the authors calculated the potential exposure risk to pesticides for different species by using different life history traits, physiology and presence/absence data of species. However, Wagner et al. (2015) only covered those species listed under Annex II of the Habitats Directive within their “Special Areas of Conservation” (SACs) (EU, 1992). Starting from this point, we decided to conduct a risk evaluation for as many reptile species as possible within their entire distribution range in the EU.

The goal of the present study was (1) to detect which taxa do generally occur within agricultural areas with regular pesticide applications and (2) to create an exposure index for all considered taxa by further taking into consideration life history and physiological traits of the species. We then (3) compared species occurring within the different pesticide admission zones within the EU (Regulation (EC) No 1107/2009) and evaluated, whether there are differences in pesticide exposure risk for each admission zone.

2. Methods

We proceeded using the method already applied by Wagner et al. (2015), albeit with slight modifications.

2.1. Identification of species occurring in agricultural areas

In this evaluation, we considered the risk of pesticide exposure of a species based on its regular presence or absence within agricultural areas. Contrary to Wagner et al. (2015), we only used literature data to determine species' occurrence within agricultural areas with regular pesticide applications (ARAs), since no pseudo-absence points for logistic regression analyses could be created at scale of the species' entire European range (for the creation of absence points in conservation areas, see Wagner et al., 2015). Data for species absence (as within SACs) was not available at the European scale. Thus, absence points within defined areas where species are known to be absent could not be generated in order to calculate logistic regressions using presence/absence of a species as predictor variable for general occurrence.

A regular occurrence within cultivated landscapes was only expected if evidence was found, i.e. multiple reports ($\geq$3) attesting the presence of individuals in agricultural landscapes, known habitat preferences of a species or visual confirmation of species in the field. In case of enough evidence of a regular presence within ARAs (i.e. when at least one of the criteria was met), 1 Risk Point (RP) was awarded (otherwise 0 RP).

2.2. Species' physiology

This evaluation considered physiological factors of a species, which should increase the potential pesticide uptake. Additionally to snout-to-vent-length (SVL), which was also evaluated by Wagner et al. (2015), we included average body mass (BM) in this evaluation. Data was retrieved from literature, and a classification scheme was established through the histogram function using the software R (R Developmental Core Team, Vienna). According to this method, SVL was classified into eight classes (0–10 cm, 10–20 cm, 20–40 cm, 40–60 cm, 60–80 cm, 80–100 cm, 100–120 cm, 120–140 cm), while BM was classified into eleven classes (0–10 g, 10–20 g, 20–40 g, 40–60, 60–80 g, 80–100 g, 100–200 g, 200–300 g, 300–400 g, 400–500 g, >500 g). The lower the BM and SVL, the higher the risk class a species was assigned to, as species with a small body size tend to exhibit a greater increase in dietary exposure when compared to larger species (Ellgehausen et al., 1980). Likewise, a small body size comes with a greater surface area, which can promote a higher dermal uptake of pesticides (Weir et al., 2010). 8 RP could subsequently be scored for SVL and 11 for BM. A total of 19 RP could thus be scored for physiology.

2.3. Species' life-history

This evaluation referred to life-history traits that may make populations of a species more or less susceptible to suffer from negative effects of pesticide exposure (mean number of clutches per year and mean clutch size). Time to reach sexual maturity – as used in Wagner et al. (2015) – could not be taken into account due to a great lack of data.

Species with a lower offspring and low clutch frequency (K-strategists) will probably suffer more from effects on individuals than r-strategists (Reznick et al., 2002). For instance, exposure concentrations for species with a low clutch frequency should be higher than for those with multiple clutches (Hopkins, 2005). A lower number of descendants will probably also lead to a decreased neonate survival, which could in turn cause decreasing population sizes (Guillette et al., 1994). The classification of clutch size followed the same pattern as for physiology, using the histogram function in R. Clutch size was then classified into 7 classes: 1–3, 3–6, 6–9, 9–12, 12–15, 15–18, >18 eggs/descendants per clutch. For the amount of clutches per year, the actual number of clutches was used (1, 2, 3, 5, 6; none of the considered species laid 4 clutches in a year). 7 and 5 RP could be scored for clutch size and amount of clutches per year respectively. A total of 12 RP could thus be scored for life-history. Data on home ranges for different species could not be considered, as reliable information is lacking for a great majority of them.

2.4. Calculation of an Exposure Risk Index (ERI) and Exposure Risk Factor (ERF)

A final ‘Exposure Risk Factor’(ERF), which results from the combination of the proportion of ARAs within a species European distribution ranges and an ‘Exposure Risk Index’ (ERI, defined by the species' scored RPs), was created for each taxon. This ERF reflects species' potential pesticide exposure risk according to habitat exposure, physiology and life history, as well as the proportion of agricultural area within its European distribution. Species could score a different amount of RPs for each of these three evaluations. In order to equally weight all three of them, RP scores were summed relative to the maximum possible score for the respective evaluation. Thus, RP scores in all evaluations were converted to a 0–10 scale, so that each evaluation factor had the same impact on the final ERF.

Based on these evaluations, a species could score a maximum amount of 32 RP. Taking the weighted measures into consideration, this resulted in a maximum of 30 RP (as there are three evaluations with a maximum score of 10 RP each). The ERF was then calculated using a modified formula from Wagner et al. (2015) under which a species habitat can score 0 to 1 points:

$$ERF = ERI \times \frac{ARA}{30} \times 100$$