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Forested headwaters mitigate pesticide effects on macroinvertebrate communities in streams: Mechanisms and quantification

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HIGHLIGHTS

- We examine the impact of pesticides and riparian forest on stream invertebrates.
- We quantify the influence of forest size and location on downstream communities.
- Forested headwaters provide source populations of pesticide-vulnerable taxa.
- Forested headwaters do not reduce pesticide-induced mortality.
- Recovery of vulnerable taxa depends on the length of the upstream forested reach.

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ABSTRACT

Pesticides impact invertebrate communities in freshwater ecosystems, leading to the loss of biodiversity and ecosystem functions. One approach to reduce such effects is to maintain uncontaminated stream reaches that can foster recovery of the impacted populations. We assessed the potential of uncontaminated forested headwaters to mitigate pesticide impact on the downstream macroinvertebrate communities in 37 streams, using the $\mathrm{SPEAR}_{\mathrm{pesticides}}$ index. Pesticide contamination was measured with runoff-triggered techniques and Chemcatcher® passive samplers. The data originated from 3 field studies conducted between 1998 and 2011. The proportion of vulnerable species decreased significantly after pesticide exposure even at low toxicity levels ($-4 < \mathrm{TU}_{\max} \leq -3$). This corresponds to pesticide concentrations down to 3–4 orders of magnitude below the LC_{50} value for standard test organisms. The toxicity of pesticides and the length of the forested reaches together explained 78% of variation in the community composition ($\mathrm{SPEAR}_{\mathrm{pesticides}}$). The proportion of vulnerable species doubled within the measured length of the forested stream section (0.2–18 km), whereas other characteristics of the forest or abiotic water parameters did not have an effect within the measured gradients. The presence of forested headwaters was not associated with reduced pesticide exposure 3 km downstream and did not reduce the loss of vulnerable taxa after exposure. Nevertheless, forested headwaters were associated with the absence of long-term pesticide effects on the macroinvertebrate community composition. We conclude that although pesticides can cause the loss of vulnerable aquatic invertebrates even at low toxicity levels, forested headwaters enhance the recovery of vulnerable species in agricultural landscapes.

1. Introduction

Pesticides are a major stressor for freshwater communities on a continental scale (Malaj et al., 2014). Despite continued efforts to avoid long-term effects of pesticides in the environment through an improved pesticide registration process in Europe (EC, 2009, 2013), field studies conducted within the last decade have

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consistently shown the effects of pesticides on stream invertebrates (Bereswill et al., 2013; Liess and von der Ohe, 2005; Rasmussen et al., 2013; Schäfer et al., 2012). Pesticide contamination of streams and rivers that exceeds toxic thresholds for invertebrates was also reported in studies from Australia (Sánchez-Bayo and Hyne, 2014; Schäfer et al., 2011) and North America (Kuo et al., 2012; Stone et al., 2014). The effects of such contamination were associated with reduced regional biodiversity (Beketov et al., 2013) and impaired ecosystem functions (Rasmussen et al., 2012b; Schäfer et al., 2007, 2012).

The main pesticide application period in Central and Northern Europe occurs between spring and midsummer (Bundschuh et al., 2014; Liess et al., 1999). After strong rainfall events pesticides enter lotic environments through field runoff and drainage systems, resulting in pulses of exposure for aquatic organisms (Liess and Schulz, 1999; Reichenberger et al., 2007). The SPEAR index (Species At Risk) for pesticides ($\text{SPEAR}_{\text{pesticides}}$) was specifically developed (Liess and von der Ohe, 2005) and validated (Beketov et al., 2009; Schäfer et al., 2007) to detect the effects of such contamination pulses on macroinvertebrate community composition. Using this index, it was shown that the effect of pesticides occurs below the thresholds predicted even by the most conservative first tier risk assessment in the EU (Liess and von der Ohe, 2005; Rasmussen et al., 2012a; Schäfer et al., 2012). Furthermore, pesticide application is predicted to increase in temperate regions due to climate change, potentially exacerbating their impact on non-target organisms (Kattwinkel et al., 2011; Noyes et al., 2009). Thus, developing mitigation measures in parallel to improving the pesticide regulation is crucial to prevent damage to the environment.

$\text{SPEAR}_{\text{pesticides}}$ has been applied to identify impact-reducing factors on a landscape level. Several previous studies have shown that upstream forested reaches (UFRs) alleviate the impact of pesticides on the macroinvertebrate community composition at downstream sites (Bunzel et al., 2014; Liess and von der Ohe, 2005; Schäfer et al., 2007; von der Ohe and Goedkoop, 2013). There was a higher proportion of pesticide-vulnerable taxa at the sites with UFR compared to the sites without UFR. However, to apply such knowledge to the development of mitigation strategies, a better understanding of the underlying mechanisms is required.

The increased proportion of pesticide-vulnerable taxa downstream of the forested reaches was mainly observed several months after contamination and was explained by enhanced recolonization processes (Liess and von der Ohe, 2005). Most previous studies did not specifically consider the influence of the UFR during the main period of pesticide application, except for Schäfer et al. (2007), who observed the influence of forested stream reaches less than 1 month after pesticide exposure. In a different study, Schäfer et al. (2012) showed that the thresholds for community-level pesticide effects were higher in streams with UFR. However, it remained unknown whether the higher threshold is related to reduced mortality of vulnerable invertebrates after pesticide exposure or enhanced recolonization. Furthermore, Harding et al. (2006) showed that forested headwaters can change physico-chemical habitat characteristics downstream. However, none of the previous studies using the SPEAR approach investigated exactly which characteristics of the forest influence the vulnerable taxa downstream of the forested reach.

Hence, the aim of our study was to investigate the influence of the UFR and associated parameters on pesticide-impacted communities. We examined the $\text{SPEAR}_{\text{pesticides}}$ values before and shortly after (<1.5 months) the measured contamination in 37 streams during the main period of pesticide application. Additionally, to identify the most important parameters associated with the UFR, we analyzed the influence of selected characteristics of the forest (length of forested reach, surface area, direct and downstream distance to sampling site) and abiotic conditions at the sampling sites on macroinvertebrate community composition, using $\text{SPEAR}_{\text{pesticides}}$.

2. Methods

2.1. Study area and sampling schedule

We compiled data on pesticide contamination, physico-chemical habitat characteristics and macroinvertebrate community composition collected in 37 streams, with one sampling site per stream. The data were collected in three field studies conducted between April and June in 1998–2000 (Liess and von der Ohe, 2005), 2010 (Münze et al., in review) and 2011 (unpublished data). From the study of Liess and von der Ohe (2005), we selected the sampling year with the highest contamination for repeatedly sampled sites. The streams were located in a crop-growing area between the cities of Braunschweig and Halle in Central Germany (Fig. 1). The dominant crops in the area were winter wheat, winter barley, sugar beets, and rape seed. Sites were selected so that no dredging occurred at least one year before or during the sampling and there were no sources of industrial or municipal effluents in the entire watercourse upstream of the investigated site. Arable land was present in the catchment upstream of all sampling sites. Twenty-three sampling sites additionally had upstream forested reaches (UFRs), whereas 17 did not. UFRs were defined as forested riparian corridors at least 100 m wide and 200 m long. They were located on average 3.3 km and a maximum of 11 km upstream of the sampling site. To ensure that UFRs were not contaminated with agricultural pesticides, they were chosen so that no arable land was present upstream of these reaches. We assessed the macroinvertebrate community composition once before the peak measured pesticide contamination and once after. That corresponded to invertebrate sampling in April/May or in May/June, respectively. However, the 5 streams from the field study in 2010 were sampled only once, after the pesticide input was measured, in June. Thus, these data were not included in the analysis involving $\text{SPEAR}_{\text{pesticides}}$ values before contamination, but were included in all other analyses.

2.2. Environmental variables

Physico-chemical parameters were measured in the field during the invertebrate sampling following the pesticide contamination. In the field investigations in 1998–2000 and in 2011, dissolved oxygen concentrations, pH and conductivity were measured using Extech dissolved oxygen and conductivity meters (DO600 and EC500 ExStik® II, Extech Instruments Corp., Waltham, USA). The phosphate and nitrate concentrations were measured using colorimetric tests by Visicolor® (Macherey&Nagel, Düren, Germany). For the study in 2010, the nutrient concentrations and conductivity measurements were obtained from the monitoring stations of the municipal water management agency (LHW Saxony-Anhalt) located near the sampling sites. In the study conducted in 2011, the nutrient concentrations were not measured (11 out of 37 sites).

2.3. Pesticide concentrations

The runoff containing the highest pesticide concentrations occurred between April and mid-June, following rainfall events of ≥10 mm/day. Pesticide contamination was quantified using runoff-triggered sampling techniques (Liess and von der Ohe, 2005) and duplicate Chemcatcher® passive samplers in the polar configuration (Stephens et al., 2005). In 1998–2000 automated runoff samplers and runoff-triggered samplers were used, in 2011 runoff-triggered and Chemcatcher® and in 2010 only Chemcatcher®. For an overview of sampling methods applied at each site, please refer to the Supplementary information (Table S1).

The runoff-triggered and automated samplers are described in detail by Liess and von der Ohe (2005). The automated sampler collected 500 mL samples every 8 min during 1 h as soon as runoff was indicated by conductivity and water level sensors (a decline in conductivity by