STATISTICAL ECO(-TOXICO)LOGY

IMPROVING THE UTILISATION OF DATA FOR ENVIRONMENTAL RISK ASSESSMENT

by

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Submitted Dissertation thesis for the partial fulfillment of the requirements for a
Doctor of Natural Sciences
Fachbereich 7: Natur- und Umweltwissenschaften
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5th December 2016

1 INTRODUCTION AND OBJECTIVES

THREATS TO FRESHWATER ECOSYSTEMS FROM CHEMICAL POLLUTION

Freshwater ecosystems, such as streams, lakes and wetlands, amount to only 0.01% of the world's water and cover only 0.8% of Earth's surface (Dudgeon et al., 2006; Gleick, 1996), yet they host an important component of global biodiversity. Freshwaters are a habitat for more than 125,000 species which represents 10% of global biodiversity and ⅓ of all vertebrate species (Balian et al., 2007; Strayer and Dudgeon, 2010) and provide essential services for human well-being (Millennium Ecosystem Assessment, 2005). Small water bodies are of particular importance because of their high abundance (Downing et al., 2012), the high biodiversity they host (Davies et al., 2008) and the ecosystem services they provide (Biggs et al., 2016).

The earth is currently experiencing a functional change driven by human activities which are so far-reaching, that a new geological epoch "Anthropocene" has been proposed (Crutzen, 2002; Steffen et al., 2011; Waters et al., 2016). Consequently, this is also associated with detrimental biotic changes: 65% of rivers are currently at threat (Vörösmarty et al., 2010), 21% of 27,516 assessed freshwater species are currently threatened with extinction (IUCN, 2016) and greatest biodiversity losses are observed in freshwater ecosystems (WWF, 2016). A multitude of stressors contribute to this deterioration of freshwater biodiversity, including habitat loss and degradation, overexploitation, invasive species and pollution (Collen et al., 2014; Dudgeon et al., 2006; Vörösmarty et al., 2010; WWF, 2016). Previous studies investigating water pollution have mainly focused on nutrient loading, acidification and pollution by organic loading (Schäfer et al., 2016). However, chemicals have become ubiquitous in mankind and are indispensable for society and economy. Currently, more than 100,000 chemicals are registered and in daily use (Schwarzenbach et al., 2010; Schwarzman and Wilson, 2009). Some of these chemicals degrade quickly, while others rather accumulate in the environment (Fenner et al., 2013).

Despite their potential negative effects on biota and their intentional release, pesticides have been neglected in the past by ecological studies investigating threats to freshwaters (Schäfer et al., 2016) and it is unknown how much they contribute to biodiversity loss (Persson et al., 2013; Rockström et al., 2009). However, recent studies indicate that pollution by pesticides may be a frequent threat to freshwaters. Malaj et al. (2014) showed that almost half of the European water bodies are at risk from pesticides. In the United States, Stone et al. (2014) showed that 61% of assessed agricultural streams exceed thresholds for a healthy aquatic-life. On a global scale, Stehle and Schulz (2015) found that 52% of detected insecticide concentrations (n = 11,300) exceeded regulatory threshold levels (RTL) and that biodiversity is reduced by ~30% at the RTL. Small streams are particularly exposed to pesticide pollution because of their large contact area with adjacent land and low water volume (Biggs et al., 2016). However, there is currently a lack of data on pesticide pollution of small streams (Lorenz et al., 2016).

As a reaction to the degradation of freshwaters, several legal frameworks have been established to safeguard and improve the quality of freshwater ecosystems. In the European Union (EU), the Water Framework Directive (WFD, European Union (2000)) regulates the protection of aquatic ecosystems and commits the member states to monitor chemical pollution and to achieve a 'good' status of all water bodies. Knowing of the toxic potential of pesticides and their intentional release into the environment, also the introduction of new pesticides is strictly regulated. Sophisticated environmental risk assessment procedures have been developed and are requested by the EU to ensure that the use of pesticides does not cause unacceptable effects to non-target organisms, soil, air and water (European Union, 2009).

ENVIRONMENTAL RISK ASSESSMENT

Environmental risk assessment (ERA) evaluates risks to animals, populations, communities or ecosystems. ERA investigates if a chemical can be used as intended without causing detrimental impacts to the environment. Therefore, ERA is also a tool to support decision making under uncertainty (Newman, 2015). Environmental risk is defined as a combination of the severity and the probability of occurrence of a potential adverse effect in the environment (Suter, 2007). Therefore, ERA is based on two components: Effect- and exposure assessment. A combination of both is needed to characterise environmental risks.

Effect assessment characterises the strength of ecological effects using laboratory, semi-field and field experiments. This is done by establishing relationships between the concentration of a compound and the observed effects. In the EU a tiered approach with increasing complexity and realism has been established. Lower tier assessment is based on highly standardised single species laboratory experiments. If a low risk cannot be established in lower tiers, higher tier assessment refines the assessment by testing additional species, extended laboratory experiments or model ecosystem experiments and aims to reduce the uncertainty in the assessment (Brock et al., 2006; EFSA, 2013). To address the various uncertainties in effect assessment (e.g. experimental variation, variation between species, variation in environmental conditions etc.) the estimated toxicity values are divided by an assessment factor (AF) between 100 (lower tier assessment) and 2 (higher tier assessment) depending on data quality, which leads to a regulatory acceptable concentration (RAC) (Brock et al., 2006; EFSA, 2013).

Exposure assessment for freshwaters aims to characterise the potential contact of the ecological entity with the chemical by deriving a predicted environmental concentration (PEC) in surface waters and sediments (Newman, 2015). This derivation is mainly based on modelling the fate of chemicals in the environment using computer simulations. In the European Union the FOCUS models are used to derive PECs (EFSA, 2013;

FOCUS, 2001). For their calculations, these models need many compound specific input parameters like the molecular weight, water solubility, partitioning coefficients and dissipation time. Additionally, information on the application regime and crop type is needed. FOCUS estimates the concentration within edge-of-field streams of 1 m width (corresponding a catchment size of approx. 5 to 40 km^2, see Figure **??**) and 30 cm depth (Erlacher and Wang, 2011). Nevertheless, recent research showed that FOCUS models fail to predict measured field concentrations of pesticides (Knäbel et al., 2014; Knäbel et al., 2012).

The final step in ERA is risk characterisation, putting together the information gained from effect and exposure assessment. Risk can be expressed in a quantitative way using the risk quotient approach: If the ratio PEC / RAC exceeds a value of 1 potential risks cannot be rebutted (EFSA, 2013; Solomon et al., 2000; Suter, 2007). Consequently, pesticides can be authorised only if the risk quotient is below 1, indicating that harmful effects are unlikely to happen.

ENVIRONMENTAL MONITORING

Widespread anthropogenic activities and the induced environmental changes have resulted in concerns about the state of the environment and have led worldwide to the development of environmental monitoring programs (Nichols and Williams, 2006). Pesticides applied on agricultural fields may enter aquatic ecosystems via diffuse sources like spray-drift, surface run-off or drainage (Carter, 2000; Liess et al., 1999; Schulz, 2004; Stehle et al., 2013), where they may have ecological effects such as the loss of sensitive species, a reduced leaf-litter breakdown rate or a decreased functional microbial richness (Liess and von der Ohe, 2005; Schäfer et al., 2007; Schäfer et al., 2012). For monitoring the progress towards the goal of a 'good' status and for assessment of the chemical status of surface waters the EU WFD established monitoring requirements for all European river basins (European Union, 2000). For chemical monitoring the WFD requires grab sampling and chemical analysis of 21 priority substances (of which 7 are pesticides) every third month and of 24 other pollutants (of which 12 are used as pesticides) every month. For these compounds environmental quality standards (EQS) have been derived that define maximum permissible concentrations (European Union, 2013). Additionally, substances that may pose a significant risk, have currently an insufficient data basis and are candidates for future priority substances can be monitored (*"watch list"*). These are currently 14 substances (of which 8 are used as pesticides, including all Neonicotinoids) that are monitored until 2019 (European Union, 2015). Nevertheless, monitoring programs on a national scale might consider a broader spectrum of chemical substances adapted to national requirements, e.g. for investigative monitoring. However, recent studies indicate that the current sampling and chemical analyses strategy greatly underestimates the pesticide exposure (Moschet et al., 2014; Stehle et al., 2013; Xing et al., 2013).

Environmental monitoring produces humongous amounts of data containing information on pesticide concentrations in the field. Moreover, data from long-term monitoring programs can be used to study hypotheses about spatial and temporal dynamics and interactions that are not evident from short term and short scale studies (Gitzen, 2012) and provide insights for modelling approaches. Therefore, it can be complementary to ERA (Suter, 2007). If the environmental risk assessment process captured all relevant sources of risk no concentrations above the derived RAC should be observable in European rivers. Therefore, monitoring data could be used to provide feedback for ERA after approval (Knauer, 2016). However, monitoring under the WFD has its main focus on large water bodies >10 km^2 catchment size (European Union, 2000), whereas ERA has its focus on small water bodies of approx. 5 to 40 km^2 catchment size (Figure **??**, Brock et al. (2006) and European Union (2009)). At present little is known on pesticide concentrations in small streams comparable to those assessed in ERA (Biggs et al., 2016; Lorenz et al., 2016).

STATISTICAL ECOTOXICOLOGY

As outlined, environmental effect assessment is based on experimental approaches and generates data on ecological effects. The produced datasets range from small univariate datasets (lower tier assessment) to medium-sized multivariate datasets (higher tier assessment). To extract usable information for assessment, these datasets are analysed using statistical techniques (Newman, 2012). Statistical ecotoxicology combines statistics with the specific needs and constraints of ecotoxicology. Ecotoxicologists deal generally with low replicated experiments, complicating statistical inference (Van Der Hoeven, 1998): A recent analysis of 11 mesocosm studies revealed that the sample sizes for these kind of experiments range between two and five (Szöcs et al., 2015). Statistical ecotoxicology aims to provide solutions to statistical challenges in ecotoxicology (Fox and Landis, 2016a), guidance on experimental designs (Johnson et al., 2015) and tools to integrate big data (Van den Brink et al., 2016). The ultimate goal is to improve the accuracy of ERA.

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The relationships between the concentration of a compound and the observed effects are usually analysed using dose-response models, which can be used to derive an effective concentration (EC_x) for x% effect (Ritz, 2010). Nevertheless, such relationships cannot always be established from experimental data. For example, mesocosm experiments are conducted to characterise effects on whole biological communities. However, because of the multivariate response and potential indirect effects between species, there is no clear dose-response relationship and no models for this kind of data currently available. Recently, Green (2016) provided examples were fitting dose-response models is problematic. In such cases, a no-observed-effect concentration (NOEC) is usually derived to quantify the toxic potential.

The NOEC is the highest tested concentration that does not lead to a statistically significant deviation from the control response and therefore relies on null hypothesis significance testing (NHST). However, the use of NOEC as a toxicity measure in environmental effect assessment has been heavily criticised in the past (Chapman et al., 1996; Fox et al., 2012; Fox and Landis, 2016b; Jager, 2012; Laskowski, 1995; Warne and van Dam, 2008). One such shortcoming is the low statistical power of NHST in common ecotoxicological experiments (Van Der Hoeven, 1998). *A priori* power calculations can provide useful guidance for choosing experimental designs (Johnson et al., 2015), but are rarely used by ecotoxicologists (Newman, 2008).

Instead of conducting experiments, toxicity could be also predicted from molecular structures using quantitative structure-activity relationships (QSAR), which are usually calculated using machine learning techniques (Breiman, 2001; Cortes-Ciriano, 2016; Murrell et al., 2015). Nevertheless, in order to improve and validate these models to give sufficient prediction accuracy more data from experiments is needed (Kühne et al., 2013). Indeed, a large amount of data is available that could be used for effect and exposure assessment. Several comprehensive databases (e.g. the US EPA ECOTOX database (U.S. EPA, 2016), the Pesticides Properties Database (Lewis et al., 2016) and ETOX (Umweltbundesamt, 2016)) provide toxicity data that could be used for effect assessment. Databases like Physprop (Howard and Meylan, 2016) and PubChem (Kim et al., 2016) provide chemical properties that are needed as input for exposure models. Monitoring data provide information on realised concentrations that could be used for validation of models and retrospective feedback to risk assessment. This "big data" can provide new information and opportunities for ERA (Dafforn et al., 2015). However, it needs to be harmonised, linked and easily accessible in order to be used effectively in ERA.

OBJECTIVES AND OUTLINE OF THE THESIS

The overall goal of this thesis was to contribute to the emerging field of statistical ecotoxicology, environmental risk assessment and environmental monitoring. The main objectives were (i) to scrutinise new methods in statistical ecotoxicology and effect assessment, (ii) explore risk dynamics using available monitoring data and (iii) provide tools to deal with and integrate big data in ERA. Figure 1.1 provides a conceptual overview on ERA and environmental monitoring as outlined in the previous sections, as well as the parts considered in this thesis and their relationships.

The thesis starts with a comparison of statistical methods to analyse ecotoxicological experiments using NHST in effect assessment (Chapter **??**). Specific questions addressed were:

- Are state-of-the-art statistical methods that explicitly consider the type of analysed data, more powerful than currently used methods for NHST?
- How much statistical power do current experimental designs in ecotoxicology exhibit?

Risk assessment procedures in the European Union have the main focus on small water bodies adjacent to agricultural fields where plant protection products are applied. Therefore, chapter **??** focuses on measured environmental concentrations on a large spatial scale in small streams, their drivers and comparison with RACs derived from ERA. Specific goals of this study were:

- Compile monitoring data on pesticides in small streams in Germany and check if the available data is suitable to inform ERA.

- Explore the relationship between agricultural land use, stream size and RAC exceedances.
- Scrutinise the annual dynamics of pesticide exposure, as well as the influence of precipitation on measured pesticide concentrations.
- Assess the current pollution in small streams using RACs from ERA and identify pesticides exhibiting currently a risk to freshwaters.

The compilation of monitoring data from different data sources in Chapter **??** resulted in a big inhomogeneous amount of data. Moreover, biologists, chemists and ecotoxicologists face similar problems with the need to identify and harmonise their biological and chemical data. Chapters **??** (chemical data) and **??** (biological data) describe software solutions to simplify and accelerate the workflow of:

- validating and harmonising chemical and taxonomic data
- linking datasets from different databases
- retrieving properties and identifiers

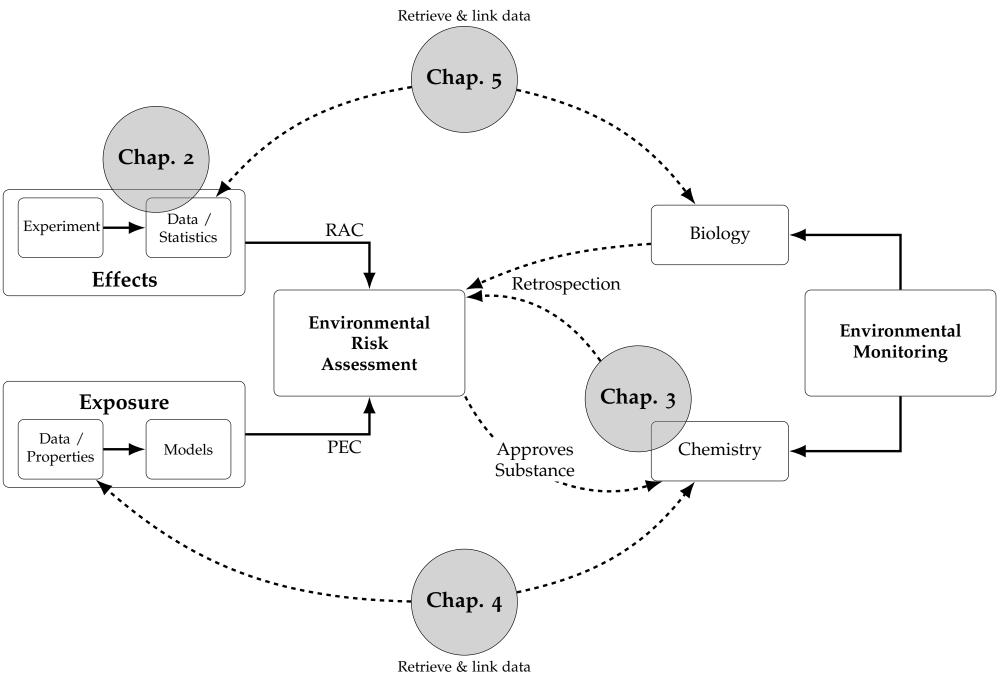

Figure 1.1: Conceptual overview on environmental risk assessment, environmental monitoring and the parts addressed by this thesis.

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