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Mapping ecological risk of agricultural pesticide runoff

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Abstract

A screening approach for the EU-scale is introduced and validated that predicts pesticide runoff and related ecological risk for aquatic communities in small agricultural streams. The approach is based on the runoff potential (RP) of stream sites, by a spatially explicit calculation based on pesticide use, precipitation, topography, land use and soil characteristics in the near-stream environment. The underlying simplified model complies with the limited availability and resolution of data at larger scales. RP is transformed to ecological risk by means of a runoff–response relationship between RP and invertebrate community composition that results from a large-scale investigation and considers the influence of landscape-mediated recovery pools. Community composition is expressed as abundance of SPEcies At Risk (SPEAR) i.e. species that are potentially affected by pesticides because of physiological sensitivity to organic pollutants and ecological traits. The SPEAR concept was applied because it provides powerful community descriptors that are independent of habitat parameters and support comparison of pesticide effects between different geographical regions. Raster maps for the EU before the 2004 enlargement indicate that ecological risk from pesticide runoff is potentially low for streams in 34% of the grid cells with non-irrigated arable land (mostly northern countries, predicted effects at ≤20% of the streams per cell). In contrast, ecological risk is very high in 19% of the grid cells (central and southern countries, predicted effects at >90% of the streams per cell). Field investigations showed that the screening approach produced appropriate estimates of ecological risk from pesticide runoff for selected regions in Finland, France and Germany.

Keywords: Runoff; Modelling; GIS; Ecological effects; Risk characterization; Landscape

1. Introduction

Diffuse pollution from agricultural sources is one major reason for deterioration of surface water quality (Cooper, 1993). Monitoring studies showed that nutrient enrichment and sedimentation (Dance and Hynes, 1980; Lenat, 1984) as well as pesticide entry (Liess and von der Ohe, 2005) can adversely affect invertebrate community composition. Among pesticides, the group of insecticides can be highly toxic to several invertebrate species, particularly within the Arthropoda (Schulz, 2004; Liess et al., 2005). Spray drift has long been assumed to be a significant source of pesticide entry, although several field studies demonstrated that the majority of pesticides applied on arable land enters via surface water runoff (Wauchope, 1978; Van der Werf, 1996), whereas losses via subsurface flow (Logan et al., 1994; Bach et al., 2000) or spray drift (Kreuger, 1998; Bach et al., 2000) are less relevant. Concerns about diffuse pesticide pollution of water bodies and potentially adverse effects on aquatic communities gave rise to current legal formulations of the European Union (EU) such as the

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Directive 91/414/EEC concerning the placing of plant protection products on the market (EEC, 1991) or the Water Framework Directive (WFD) (EC, 2000) that presents the concept for the sustainable use of water resources by integrated river basin management. One of the environmental objectives of the WFD is to achieve and maintain a good chemical and ecological status of surface water bodies and the first step towards this is to locate water bodies that are likely not to meet this criterion. For such evaluations, water monitoring programs are indispensable that operate at the river basin scale and even larger scales within the EU. In addition, large-scale screening models that interface with Geographic Information Systems (GIS) can be helpful, as they allow an initial quick and cost-effective identification of areas of concern and provide information that can help to target regional monitoring programs more efficiently. Some studies complied with the need for addressing pesticide pollution issues EU-wide (Tiktak et al., 2004; Finizio et al., 2005), but the suggested models focused on exposure characterization, while the ecological risk resulting from pesticide exposure was not addressed. To characterize ecological risk of pesticide runoff EU-wide, a combination of information about runoff inputs and runoff–response relationships is required at broader scales. Patterns in potential runoff inputs may be simulated using mathematical models, but model complexity needs to comply with the scale of interest (Addiscott and Mirza, 1998). Owing to the limited availability and low resolution of required input data at broader scales (Steinhardt and Volk, 2003), only simple exposure models have been employed in the past years (Dabrowski et al., 2002; Verro et al., 2002; Probst et al., 2005b; Schriever et al., 2007a). Reported field effects of pesticides are valuable to set up exposure–response relationships, but often the related investigations are not of sufficient scale, and only a few studies have addressed both the effects of pollutants and the potential for recovery under field conditions (Liess and von der Ohe, 2005). Large-scale investigations (Clements et al., 2000; Potter et al., 2005; Probst et al., 2005b; Posthuma and De Zwart, 2006) are scarce for all types of pressures on aquatic systems. This paucity can be attributed to the challenge of inferring causality of exposure and biological impairment (Suter et al., 2002) in the presence of confounding factors and natural variability (Liess et al., 2005). Pattern recognition against the background of natural variability is indispensable for inferring causality and biological indicator systems (e.g. for organic pollution (Kolkwitz, 1950) or acidification (Braukmann and Biss, 2004)) can provide valuable tools. Recently, a system was developed to indicate pesticide pollution in streams (Liess and von der Ohe, 2005) by means of the concept of SPEcies At Risk. This concept classifies invertebrate species as at risk (SPEAR) or not at risk of being affected by pesticides with respect to species' traits (i.e., physiological sensitivity to organic pollutants including pesticides, generation time, migration ability and aquatic stages during the main period of agrochemical application). The concept provides powerful descriptors of community composition that are sensitive to gradients in agricultural intensity (Schriever et al., 2007a) and pesticide exposure (Liess and von der Ohe, 2005) but independent of habitat parameters, which facilitates comparison of pesticide effects between different geographical regions.

The present work introduces a screening model that was designed to characterize the ecological risk of pesticide runoff at the EU-scale, i.e. to indicate the extent to which agricultural runoff poses a threat to invertebrate communities in small streams draining arable land. The screening model is based on a simplified spatial runoff model to comply with the limited availability of input data at large scales, and uses a SPEAR-based runoff–response relationship to predict the ecological effects of runoff with respect to landscape-mediated recovery pools. Predicted aquatic risk (i.e., the predicted percentage of sites where community composition is affected due to runoff inputs and lack of recovery pools) is validated for selected regions in Finland, France and Germany.

2. Materials and methods

2.1. Runoff potential

2.1.1. Model

A generic indicator termed runoff potential (RP) (Schriever et al., 2007b) was used to distinguish stream sites with respect to the potential for runoff inputs due to key environmental characteristics of the near-stream environment (i.e., land use, topography, soil characteristics, and precipitation). The RP is based on a mathematical model that describes runoff losses of a compound with generalized properties and which was developed from a proposal by the Organisation for Economic Co-operation and Development (OECD) for estimating dissolved runoff inputs of a pesticide into surface waters (OECD, 1998). The RP was recently validated with pesticide runoff measured in small streams draining agricultural land (Schriever et al., 2007b) and was found to correlate with observed patterns in macroinvertebrate communities in small agricultural streams (Schriever et al., 2007a).

The runoff model underlying RP (Eq. (1)) calculates the dissolved amount of a generic substance that was applied in the near environment of a stream site and that is expected to reach the stream site during one rainfall event (gLOAD [g]). The dissolved amount gLOAD results from a single application in the near-stream environment (i.e., a two-sided 100-m stream corridor extending for 1500 m upstream of the site) and is the amount of applied substance in the designated corridor reduced due to the influence of the site-specific key environmental factors precipitation, soil characteristics, topography, and plant interception:

$$\mathrm{gLOAD} = \sum_{i=1}^{n} \sum_{j=1}^{m} A_{i,j} \cdot D_{\mathrm{generic}} \cdot \left(1 - \frac{I_j}{100}\right) \cdot \frac{1}{1 + \frac{\mathrm{Koc}_{\mathrm{generic}} \cdot \mathrm{OC}_i}{100}} \cdot f(s_i) \cdot \frac{f(P_i, T_i)}{P_i} \tag{1}$$

where

$A_{i,j}$ is the patch size of arable land within the stream corridor [ha],
index i refers to different patches of arable land,
index j refers to different crops cultivated,
D_{generic} is the country and crop-specific application rate of the generic substance,
I_j is the crop- and growth phase-specific plant interception at the time of the rainfall event [%]
$\mathrm{Koc}_{\mathrm{generic}}$ is the sorption coefficient to organic carbon of the generic compound, and is set to a value of 100 in order to maximize distinction of sites due to differences in soil organic carbon content,
OC_i is the soil organic carbon content of a patch [%],
s_i is the mean slope of a patch [%],
$f(s_i)$ describes the influence of slope according to Beinat and van der Berg (OECD, 1998),

$$= \begin{cases} 0.001423 \cdot s_i^2 + 0.02153 \cdot s_i, & \text{if} \quad s_i \leq 20\% \\ 1, & \text{if} \quad s_i > 20\% \end{cases} \tag{2}$$

P_i is the precipitation depth [mm],
T_i gives the soil texture of the patch (Sand/Loam),
$f(P,T_i)$ is the volume of surface runoff [mm], described according to results of Lutz and Maniak for vegetated dry soils typical of middle and late vegetation period (OECD, 1998),

$$= \begin{cases} -5.86 \cdot 10^{-6} \cdot P_i^3 + 2.63 \cdot 10^{-3} \cdot P_i^2 - 1.14 \cdot 10^{-2} \cdot P_i - 1.64 \cdot 10^{-2}, & \text{if } T_i = \text{Sand} \\ -9.04 \cdot 10^{-6} \cdot P_i^3 + 4.04 \cdot 10^{-3} \cdot P_i^2 + 4.16 \cdot 10^{-3} \cdot P_i - 6.11 \cdot 10^{-2}, & \text{if } T_i = \text{Loam} \end{cases}. \tag{3}$$

The near upstream environment of a stream site comprises a corridor area of 0.30 km^2 (2×100 m×1500 m) unless the watercourse is branched within the upstream distance of 1500 m. In the case of a branched watercourse, the near-stream environment comprises all of the corridor area along the different branches within the upstream distance of 1500 m, resulting in a corridor area of greater than 0.30 km^2. When there is one bifurcation within 1500 m upstream, the modus of the resulting corridor area of a stream site would be 0.45 km^2. Runoff losses of the generic substance (gLOAD) are predicted for each rainfall event during the main period of pesticide application in a study area. Rainfall intensity is not considered for simplification which is in accordance with the OECD model. The RP of a stream site is calculated on an annual basis according to Eq. (4):

$$\mathrm{RP} = \log(\max_{i=1}^{n}(\mathrm{gLOAD}_i)) \tag{4}$$

where

RP is the log-transformed maximum of n gLOAD values,
n is the number of rainfall events that occur during the main period of pesticide application,
gLOAD_i is the amount of a generic substance that potentially reaches a stream site during rainfall event i as given in Eq. (1).

RP values are log-transformed in order to classify modeled runoff losses broadly into order-of-magnitude categories.

2.1.2. Model parameterization

RP was modeled for the area of the EU before the 2004 enlargement (EU-15). Spatial input data of 10 km and 50 km resolution only were available at this scale (Table 1). Therefore, RP was not calculated for the near-stream environment of single stream sections but was modeled per grid cell. This means that gLOAD (Eq. (1)) was calculated from the amount of arable land in a given grid cell (x,y) and the rule of proportion was applied as shown in Eq. (5) to obtain the theoretical amount of arable land in the near upstream environment of a stream site located in the grid cell (x,y):

$$A_{\mathrm{stream}\ i,j} = A_{\mathrm{cell}\ i,j} \frac{E_{\mathrm{stream}\ i,j}}{E_{\mathrm{cell}}} \tag{5}$$

where

$A_{\mathrm{stream}\ i,j}$ is the amount of arable land in the near upstream environment of a stream site located in grid cell (x,y)

Table 1
Input data and sources [a]

Data	Type of data	Resolution	Source
Use of plant-protection products (1999)	Tabulated	Country	EUROSTAT
Crop area (FSS 2000)	Tabulated	NUTS III	EU-JRC, IES
MARS grid	Raster map	50-km grid	EU-JRC, IES
Daily recorded precipitation (2000)	Tabulated	50-km grid	EU-JRC, IES
CORINE land cover data (2000)	Raster map	10-km grid	EU-JRC, IES
Slope	Raster map	10-km grid	EU-JRC, IES
Soil texture	Raster map	10-km grid	EU-JRC, IES
Soil organic carbon content	Raster map	10-km grid	EU-JRC, IES
Hydrological network	Shape file	250-m grid	EU-JRC, IES
Catchments	Shape file	250-m grid	EU-JRC, IES

[a] EUROSTAT: Statistical Office of the European Commission; EU-JRC, IES: European Commission Joint Research Centre, Institute for Environment and Sustainability; FSS: Farm Structure Survey; NUTS: Nomenclature of territorial units for statistics; CORINE: Coordination of Information on the Environment; MARS: Monitoring Agriculture with Remote Sensing.

$A_{\mathrm{cell}\ i,j}$ is the amount of arable land in cell (x,y)
$E_{\mathrm{stream}\ i,j}$ is the theoretical size of the near-stream environment of the stream site located in grid cell (x,y); equal to 0.45 km^2, i.e. the modus size of the near-stream environment in the case of one bifurcation of the upstream watercourse
E_{cell} is the size of the grid cell (x,y) equal to 100 km^2

As a result, gLOAD reflected the mean generic exposure of a stream section, which is located in cell (x, y) and has the same environmental characteristics as the grid cell (including percentage of arable land). The amount of arable land per grid cell was specified according to CORINE land cover data (Table 1). The CORINE land cover class 2.1.1 "Non-irrigated arable land" was considered exclusively because modeled runoff inputs were to be linked with a runoff–response relationship (described below) that was established in an agricultural area where this type of arable land is predominant. Non-irrigated arable land includes fields under crop rotation that may be sporadically irrigated using non-permanent infrastructure (sprinkler irrigation). The class does not include permanent pasture and crops that are irrigated permanently or periodically, using a permanent infrastructure (irrigation channels, drainage network). The latter is defined as CORINE land cover class "Permanently irrigated land" (http://reports.eea.europa.eu/COR0-landcover/en). Permanently irrigated arable land is abundant in Greece, Portugal and Spain (20% of arable land, details on distribution in these countries further down) but of no relevance in the other EU-15 countries (0%).

The calculation of gLOAD was based on shares of cereals (largest share on average), grain maize, potatoes, sugar beets, rape and vegetables in arable land per grid cell (calculated from Farm Structure Survey statistics, Table 1), because the majority of pesticide application in the EU is associated with these crops (EUROSTAT, 2002). The application rate D_{generic} reflected relative differences in pesticide use between the EU-15 countries and was specified per crop and country in two steps. Firstly, the cumulative volumes of all pesticides applied per crop and country were divided by the amount of crop area per country (Fig. 1). Secondly, the resulting application rates were rescaled to arrive at values of gLOAD and RP, respectively, which could be linked to the designated runoff–response relationship (described below). Since a generic application rate of 1 g ha^{-1} for all major crops was inherent to that relationship, the crop-specific application rates for the EU-15 countries were rescaled in that they were divided by the application rates for Germany. Values of D_{generic} were specified according to data from 1999 since data from 2000 were not available. The consequences of combining application data from 1999 with other environmental data from 2000 can be considered as minor, since the data originate from surveys in subsequent years and cumulative application rates are unlikely to change drastically from one year to the next.

The plant interception of each major crop under cool and warm climate conditions was specified according to values published in Huber et al. (2000). For cereals, differences in plant interception between cool and warm climates were small during the months of May to July. Information on slope as well as on the texture and organic carbon content of soils was extracted from the data sources listed in Table 1. All spatial data processing used the GIS Arc View 3.2a (ESRI, Redlands, CA, USA).

Runoff losses of the generic substance (gLOAD) were predicted for each day with recorded precipitation during the period April to July in the year 2000 (latest available precipitation data, Table 1). Daily recorded

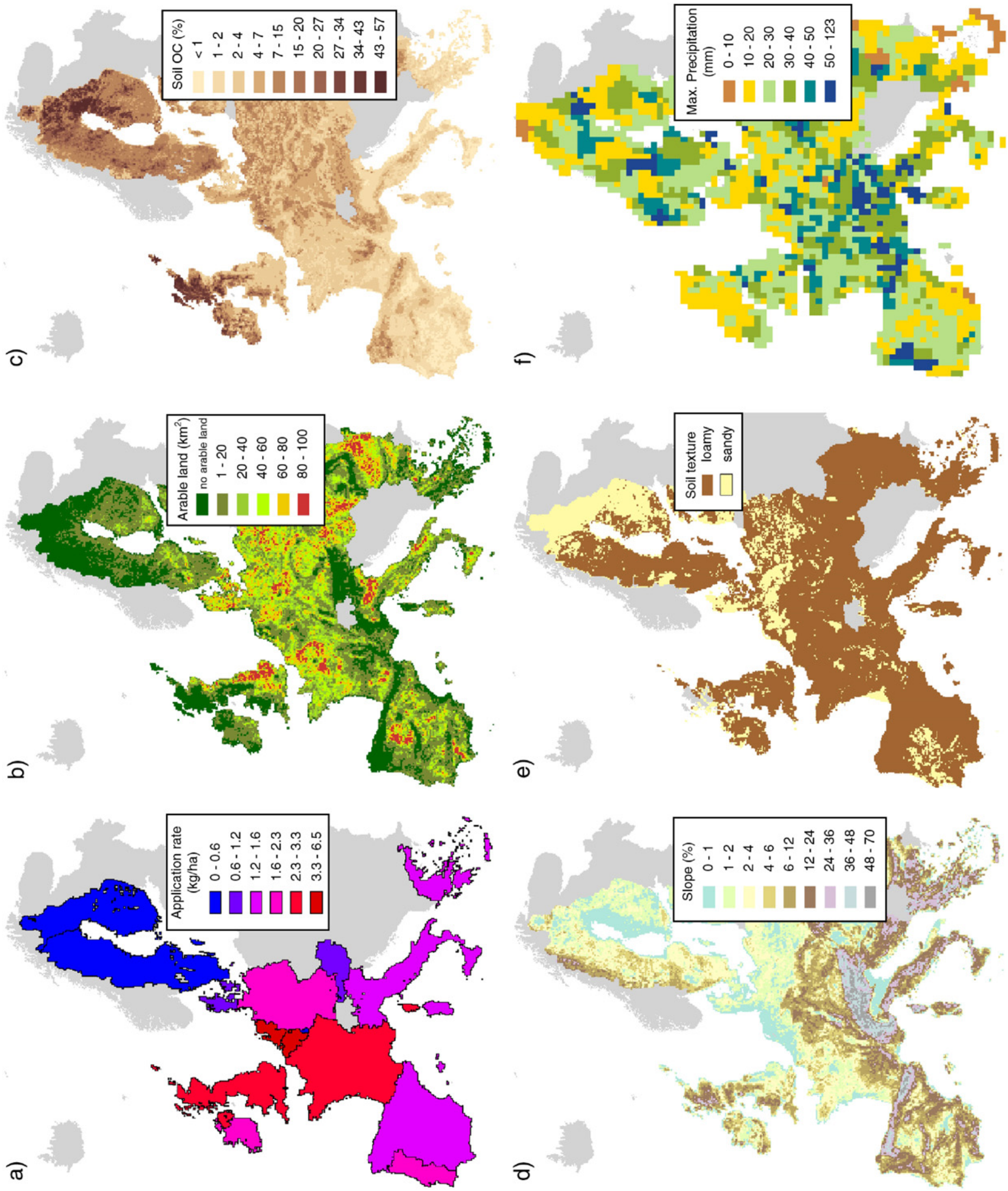
a)
Application rate (kg/ha)
0 - 0.6
0.6 - 1.2
1.2 - 1.6
1.6 - 2.3
2.3 - 3.3
3.3 - 6.5
b)
Arable land (km²)
no arable land
1 - 20
20 - 40
40 - 60
60 - 80
80 - 100
c)
Soil OC (%)
< 1
1 - 2
2 - 4
4 - 7
7 - 15
15 - 20
20 - 27
27 - 34
34- 43
43 - 57
d)
Slope (%)
0 - 1
1 - 2
2 - 4
4 - 6
6 - 12
12 - 24
24 - 36
36 - 48
48 - 70
e)
Soil texture
loamy
sandy
f)
Max. Precipitation (mm)
0 - 10
10 - 20
20 - 30
30 - 40
40 - 50
50 - 123

Table 2
Predicted ecological risk for streams in cell (x,y) due to runoff potential (RP) and forested upstream reaches that provide recolonization pools along hydrological networks

Level of runoff inputs intostream sites in cell (x,y)				Ecological risk for streams in cell (x,y): percentage of streams sites potentially affected	
RP	RP class	Effect value [a]	Percentage of sites in cell (x,y) with forested upstream reaches	Median risk$_{\mathrm{cell}(x,y)}$ (lower; upper estimate)	Risk class
≤−5	Very low [b]	0	0–100%	0 (0; 0)%	Very low
≤−3	Low	0	0–100%	0 (0; 0)%	Low
>−3	Medium	0.5	≥50%	10 (0; 20)%	Low
>−3	Medium	0.5	<50%	37.5 (>20; 55)%	Medium
>−2	High to Very high	1***	≥50%	37.5 (>20; 55)%	Medium
>−2	High to Very high	1***	<50%	72.5 (>55; 90)%	High
>−2	High to Very high	1***	<10%	72.5 (>55; 90)%	High
>−2	High to Very high	1***	<10%	95 (>90; 100)%	Very high

[a] Long-term shifts in community composition corresponding to RP, *** $p<0.001$, for details see Materials and methods section.
[b] Non-irrigated arable land is not indicated according to CORINE data.

precipitation was assumed to result from one rainfall event, because the data did not include information about single precipitation events per day. Maximum daily rainfall occurred during the months of May to July. The period from April to July was considered, because for the major crops it represents the main application period of insecticides (Huber et al., 2000), i.e. the group of pesticides with the largest potential to adversely affect invertebrate species (Schulz, 2004; Liess et al., 2005). Runoff losses (gLOAD) were predicted to be 0, when CORINE data (minimum mapping unit of 25 ha) did not indicate non-irrigated arable land in a grid cell. In these cases, a generic value of gLOAD was assigned to indicate potential but minimum exposure, which was set equal to half of the smallest gLOAD calculated from CORINE data on non-irrigated arable land. The RP per grid cell was calculated from the set of gLOAD data according to Eq. (4) and was binned in five order-of-magnitude categories that ranged from very low to very high as shown in Table 2. The RP value of a grid cell reflected the mean runoff inputs into a stream that is located in the grid cell with the specified key environmental factors.

2.1.3. Sensitivity analysis

Uncertainties related to input data will propagate to model output and will increase the level of uncertainty relating to output data. Therefore, it is necessary to know the robustness of model outcomes and how they relate to changes in each of the model parameters. For this purpose, model sensitivity was assessed in detail for gLOAD by means of the sensitivity index S (Jorgensen, 1995). The index (Eq. (6)) relates changes in parameter values to changes in model outcomes, which is done with respect to the ratio of absolute parameter values to absolute model outcomes:

$$S_{\mathrm{gLOAD}}(K) = \left|\frac{\partial\,\mathrm{gLOAD}}{\partial K}\right|_{K=K_i} \cdot \frac{K}{\mathrm{gLOAD}(K)} \qquad (6)$$

where

K is the considered parameter and
gLOAD(K) is the model dependent on K.

Positive values of S_{gLOAD} indicate that if parameter values increase, also gLOAD values increase. Negative values of S_{gLOAD} indicate that if parameter values increase, gLOAD values decrease. The larger $|S_{\mathrm{gLOAD}}|$, the more the model is sensitive to changes in parameter K, i.e. the more gLOAD changes as parameter values change. If $|S_{\mathrm{gLOAD}}|$ is a function of parameter K, then the influence of parameter changes on changes in gLOAD varies across the parameter range. If $|S_{\mathrm{gLOAD}}|$ is a constant of 1, then the influence of parameter changes on changes in gLOAD is unchanged across the parameter range. The sensitivity index was determined by varying one parameter in range, while all other parameters were kept constant at mean values for the EU-15 area (precipitation = 30 mm, slope = 8.1%, arable land = 3076 ha and soil organic carbon content = 6.9%) or at specified values (generic (standardized) application

Fig. 1. Raster maps of selected input data for predicting runoff potential (RP). a) application rate, b) amount of arable land, c) topsoil organic carbon content, d) slope, e) soil texture, f) maximum values of daily precipitation (April to July 2000). Data from the Statistical Office of the European Commission (EUROSTAT) and European Commission Joint Research Centre, Institute for Environment and Sustainability (EU-JRC, IES).

rate = 1; $Koc_{generic}$ = 100). Mean values of plant interception reflected the average situation for the major crops in the month of May (65–80% plant interception for cereals and rape, 5–15% plant interception for grain maize, potatoes, sugar beets, and vegetables). The parameters were varied in range as follows: generic (standardized) application rate was varied across the maximum parameter range for the EU-15 area (0.24 g ha^{-1} to 3.10 g ha^{-1}). Mean values of precipitation, slope, arable land, soil organic carbon content and plant interception were varied in range by +/−25%. All computing was done with the software Mathematica 4 (Wolfram Research, Champaign, IL, USA).

3. Spatial link between runoff potential and community composition

Modeled RP was studied recently for small streams draining arable land in the region of Braunschweig, Central–North German Lowlands. A significant positive relationship was found between RP and measured runoff inputs (Schriever et al., 2007b). Furthermore, a spatial link was established between RP and long-term shifts in the composition of benthic invertebrate communities expressed as percentage of SPEAR abundance (%SPEAR abundance) (Schriever et al., 2007a). Another study (Liess and von der Ohe, 2005) inferred causality of long-term shifts in % SPEAR abundance and measured pesticide contamination for the Braunschweig region and showed that the relationship between measured pesticide contamination and community composition was influenced by the presence of forested upstream reaches, which can promote recovery from pesticide impacts via recolonization. The studies of Liess and von der Ohe (2005) and Schriever et al. (2007a) were the first to present field-based runoff–response relationships for the large-scale effects of agricultural intensity and pesticide exposure that additionally included the influence of landscape-mediated recovery pools.

The runoff–response relationship between RP and % SPEAR abundance was used to predict the effects of RP in small agricultural streams for the EU-15 area and is briefly described in the following (for details see Schriever et al. (2007a)): RP was modeled for 360 stream sites in the Braunschweig region from spatial data provided by regional authorities for surveying and mapping (Landvermessung und Geobasisinformation Niedersachsen, Hannover, Germany), for soil sciences (Niedersächsisches Landesamt für Bodenforschung, Hannover, Germany) and by the German Meteorological Service (Deutscher Wetterdienst, Offenbach am Main, Germany). Community composition (%SPEAR abundance) was calculated from bio-monitoring data from a 17-year period that were obtained from the regional agency for water management, coastal protection and nature conservation (Niedersächsischer Landesbetrieb für Wasserwirtschaft, Küsten-und Naturschutz, Hildesheim, Germany). Community composition was compared between different levels of RP and between two groups of sites: sites with forested upstream reaches and sites without those reaches (Kruskal–Wallis analysis-of-variance). Sites were considered to have forested upstream reaches when digital data on land use indicated that forest covered more than 20% of the near-stream environment (i.e., of a two-sided 100 m corridor on the watercourse extending for 1500 m upstream of the stream site). In the absence of forested upstream stretches, a significant shift in community composition occurred at high RP (RP > −2), where median %SPEAR abundance was decreased ($p < 0.001$) in comparison to sites of low modeled RP (RP < −3). At sites characterized by medium RP, ranging from greater −3 to −2, the decrease in % SPEAR abundance was distinct but not significant. Downstream of forested reaches, no such significant long-term decrease was observed. Even at sites of high RP, observed median %SPEAR abundance remained significantly enhanced in comparison to sites without forested reaches (Mann–Whitney *U*-test, $p < 0.01$).

4. Ecological risk of runoff

The runoff–response relationship described above allows prediction of whether modeled runoff inputs affect community composition in small agricultural streams at sites without forested upstream reaches. This means that RP can be transformed into the estimated median effect value of a grid cell. Information about the frequency of stream sites without forested upstream stretches give the potential effect frequency per grid cell, i.e. the frequency of sites where community composition could be affected if RP was high enough. The median effect value multiplied by the effect frequency yields an estimate of the ecological risk in a grid cell (Eq. (7)), i.e. of the percentage of stream sites per cell where community composition is affected due to high RP and lack of forested upstream stretches:

$$\mathrm{Risk}_{\mathrm{cell}(x,y)} = 100 \cdot \begin{cases} 0 \quad , & \text{if predicted} \quad \mathrm{RP} \le -3 \\ 0.5 \quad \cdot x_i, & \text{if predicted} \quad \mathrm{RP} \le -2 \\ 1 \quad \cdot x_i, & \text{if predicted} \quad \mathrm{RP} > -2 \end{cases} \tag{7}$$

where

0, 0.5, and 1 are effect values resulting from predicted RP for a stream site in cell (x,y),

x_i gives the probability that stream sites in cell (x, y) are without forested upstream stretches, and
index i is the percentage of forest in cell (x,y).

The effect values relating to RP levels of grid cells were derived from the runoff–response relationship between RP and %SPEAR abundance that is described in the Materials and methods section and in Schriever et al. (2007 a,b). An effect value of 0 was assigned to grid cells with RP in the range of −3 and less, because the runoff–response relationship indicated no effects at this RP level. The relationship indicated distinct (but not significant) changes in community composition, when RP values ranged from greater −3 to −2. Therefore, an effect value of 0.5 was assigned to grid cells with this RP level to account for the probability of type II errors. An effect value of 1 was assigned when the RP of grid cells was greater than −2, because the runoff–response relationship indicated significant effects at this RP level. The levels of predicted ecological risk are summarized in Table 2.

The probability that stream sites in grid cell (x,y) had no forested upstream stretches was determined by a distributional analysis of the 360 streams investigated in the Braunschweig region, Central–North German Lowlands (Schriever et al., 2007a). A 10-km grid was applied to that region in order to relate per grid cell the relative forest cover to the relative number of sites that had no forested stretches upstream. The percentage of sites without forested upstream stretches decreased as the percentage of forest per cell increased. In cells with <10% forest cover, 98% of the stream sites had no forested stretches, while in cells with ≥50% forest cover only 25% of the stream sites were lacking forested stretches.

4.1. Comparison of predicted and observed ecological risk

Predicted ecological risk of pesticide runoff (i.e., the percentage of stream sites per grid cell where community composition is affected due to high RP and lack of forested upstream stretches) was evaluated with reference to field observations from selected regions in Finland, France, and Germany. The observations were made during three field studies on the effects of pesticide exposure on benthic invertebrate communities in small streams draining arable land (Liess and von der Ohe, 2005; Schäfer et al., 2007). The level of pesticide use in agriculture differs between the three countries (Fig. 1a). Finland is characterized by the lowest pesticide use among the EU-15 countries (cumulative application rate of pesticides: 0.5 kg ha^{-1} in 1999). The cumulative application rate for Germany (2.1 kg ha^{-1} in 1999) is increased by a factor of 4 in comparison to Finland, while for France the cumulative application rate (3.3 kg ha^{-1} in 1999) is increased by a factor of 6 in comparison to Finland and by a factor of 1.4 in comparison to Germany.

Predicted ecological risk for the three study regions was calculated according to Eq. (8) considering the grid cells in which investigated stream sites are located (Liess and von der Ohe, 2005; Schäfer et al., 2007). Eq. (8) yielded the median (lower; upper) estimate of the number of sites per study region that were expected to be affected (y):

$$y = \sum_{i=1}^{m=5} m\mathrm{Risk}_{\mathrm{cell}(x,y)} \cdot n_i \qquad (8)$$

where

index i refers to one of the ecological risk classes “Very low” to “Very high”
$m\mathrm{Risk}_{\mathrm{cell}(x,y)}$ is the median (lower; upper) estimate of ecological risk for streams in grid cells that belong to risk class i (Table 2)
n is the number of investigated sites that are located in grid cells of risk class i.

The estimates were then compared with the observed number of sites per study region where community composition was affected. When the observed number of sites per study region fell into the predicted range of the number of affected sites the estimated risk was judged to correspond to observed risk. The observed number of affected sites per study region was calculated from the number of investigated sites where measured pesticide contamination was medium to high, i.e. concentrations were more than 1:10,000 of the acute median lethal concentration for *Daphnia magna* (48 h–$LC50_D$). A site with medium to high contamination was judged to be affected when %SPEAR abundance was lower than mean %SPEAR abundance (95% confidence) for moderately to highly contaminated sites in the studied region (provided that mean %SPEAR abundance at sites with medium to high contamination was significantly lower than mean %SPEAR abundance at minimally impaired sites in the studied region, independent samples t-test).

The field study in southern Finland was conducted in 2005 on 13 streams in the agricultural area between Porvoo and Turku (Schäfer et al., 2007). The study included monitoring of pesticide levels (passive water sampling) and macroinvertebrate communities in the months of July and August. During these two months, measured pesticide concentrations (only fungicide

Trifluralin detected) were five orders of magnitude below the 48 h–$LC50_D$, and %SPEAR abundance did not change significantly from July to August.

The field study in northern Germany was conducted on 20 agricultural streams in the Braunschweig region; these are independent of the 360 stream sites in the Braunschweig region described above that were investigated to link RP and %SPEAR abundance. Runoff-triggered pesticide measurements and macroinvertebrate samples were taken over three years (1998–2000) in the months of April to June and a firm link was established in that study between measured pesticide concentrations and significant reduction in %SPEAR abundance (for details see Liess and von der Ohe (2005)). Five of the investigated sites were characterized by maximum pesticide concentrations that were more than four orders of magnitude below the 48 h–$LC50_D$. For the other 15 sites, measured concentrations were in the range from 1:10,000 to 1:5 of the 48 h–$LC50_D$. Mean %SPEAR abundance for the five minimally contaminated sites (44%) was significantly higher than the respective value for the 15 sites (25%). At 11 out of the 15 sites, % SPEAR abundance was below the mean %SPEAR abundance (95% confidence) of 32%.

The field study in western France was conducted in 2005 on 16 agricultural streams in the catchments of the rivers Scorff and Ille in Brittany. The study included runoff-triggered pesticide measurements and macroinvertebrate monitoring in the months of April and May and demonstrated a link between pesticide exposure and invertebrate community composition (for details see Schäfer et al. (2007)). Maximum measured pesticide concentrations were less than 1:10,000 of the 48 h–$LC50_D$ at five sites and ranged from 1:10,000 up to 1:10 of the 48 h–$LC50_D$ at the other 11 sites. Mean %SPEAR abundance for the five minimally contaminated sites (53%) was significantly higher than the respective value for the 11 sites (26%). At 9 out of the 11 sites, %SPEAR abundance was below the mean %SPEAR abundance (95% confidence) of 36%.

5. Results

5.1. Predicted runoff potential

Predicted RP is mapped for the EU-15 countries in Fig. 2. The map indicates where agricultural activities on non-irrigated arable land can result in high pesticide runoff into adjacent streams. A general view of the map and the maps of input data presented in Fig. 1 shows the influence of different key environmental factors on RP. For instance, RP of streams was very low in the mountainous areas (e.g. the Alps in the southern EU-15 countries) due to a lack of arable land. RP of streams ranged from low to medium for 62% of the grid cells with arable land (low = 30%; medium = 32%). RP was low e.g. in the southern regions of Sweden and Finland, where small amounts of arable land are combined with slight slopes and low application rates. Despite large amounts of arable land and high application rates, RP of streams was low to medium in the northern parts of Germany, which can be attributed to low slopes and sandy soils. In the Po delta south of the Italian Alps, the amount of arable land was higher than in the northern parts of Germany, but still RP was medium due to low slopes. RP of streams was characterized as high for 36% of the grid cells with arable land. RP was high, e.g. in the southern parts of Germany and in large parts of France because here there are considerable amounts of arable land combined with steep slopes in Germany. In Brittany, in the North-west of France, RP was high because moderate amounts of arable land are combined with steep slopes. RP of streams was high in northwestern parts of Spain due to large amounts of arable land that are combined with low soil organic carbon contents. RP was characterized as very high for only 2% of the grid cells with arable land (e.g., north of the French Pyrenees and in southeastern Italy) where high amounts of arable land on predominantly loamy soils are associated with high maximum daily rainfall. Differences in RP due to plant interception could be expected between cool and warm climate regions. However, in the period of May to July (i.e., during the months of maximum daily rainfall), differences in plant interception between these regions were small, especially for cereals, which on average are the major crop per district. Therefore, differences in RP between grid cells due to plant interception were small in comparison to the influence of the other environmental factors.

5.2. Sensitivity of the runoff model

The sensitivity index S_{gLOAD} was calculated for the parameters precipitation, slope, amount of arable land, soil organic carbon content and plant interception by varying mean parameter values in range by +/−25%. All indices were dependent on absolute parameter values except for S_{gLOAD} (amount of arable land) and S_{gLOAD} (generic application rate), each of which was a constant equal to 1 over the whole parameter range (Fig. 3a). Positive values were calculated for the indices S_{gLOAD} (amount of arable land), S_{gLOAD} (precipitation), and S_{gLOAD} (slope), while negative index values were calculated for S_{gLOAD} (plant interception) and S_{gLOAD} (soil

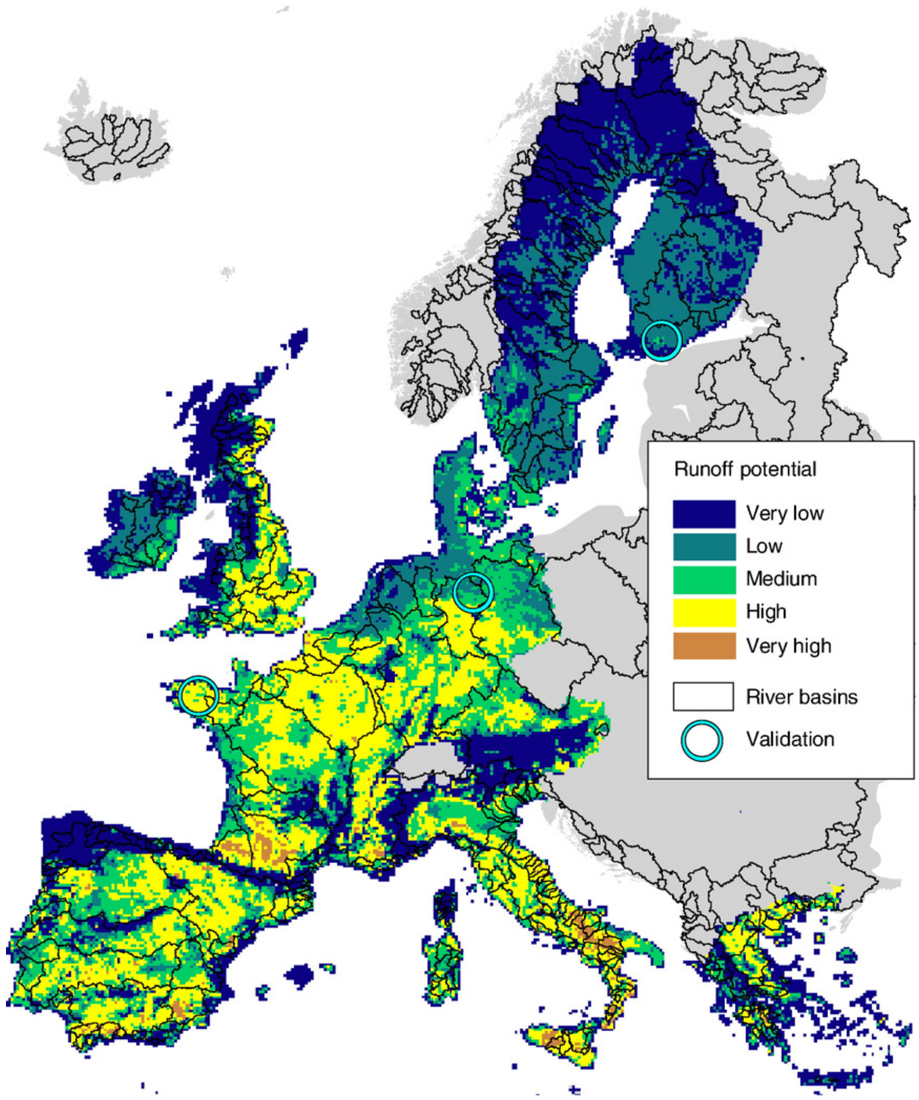

Fig. 2. Distribution of predicted runoff potential (RP) in the EU-15 countries (10-km grid). For details on the RP classification see Table 2.

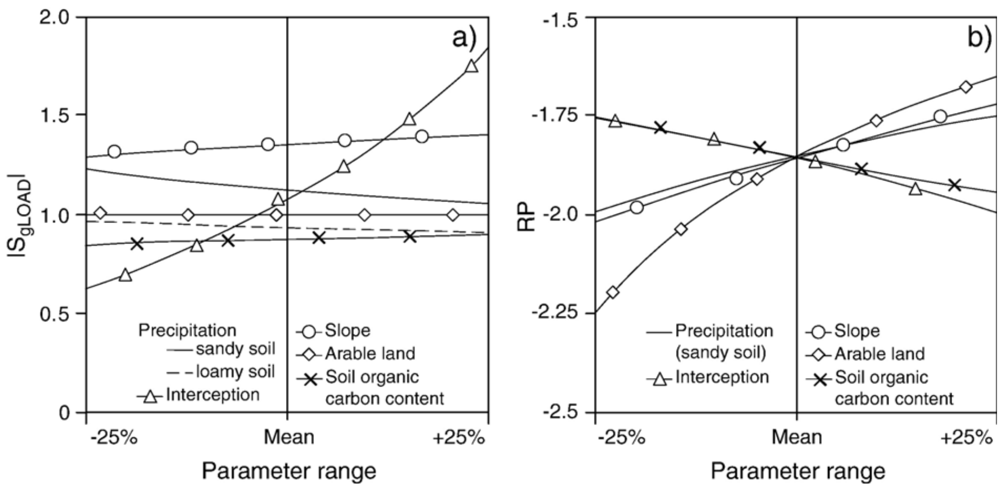

Fig. 3. Results of the sensitivity analysis. a) Influence of parameter changes on modeled gLOAD. b) Influence of parameter changes on modeled runoff potential (RP).

organic carbon content). Values of $|S_{\mathrm{gLOAD}}|$ were highest for the parameters plant interception, slope, and precipitation, i.e. the uncertainty related to input data for specifying these parameters can strongly influence the level of uncertainty related to gLOAD.

The influence of parameter changes on RP is demonstrated in Fig. 3b. The RP value of a grid cell that is characterized by mean values of environmental parameters and an application rate of 1 g ha^{-1} is -1.85 corresponding to RP class "High". Varying parameter values by +/−25% to reflect the uncertainty relating to the input data resulted in a change in RP by |0.10| to |0.20| with the largest change due to variation in the parameters amount of arable land (|0.20|), slope (|0.15|) and precipitation (|0.13|). Provided an RP value of −1.85, a decrease by 0.20 resulted in an RP value of −2.05, which was smaller than the lower limit of RP class "High". This means that with respect to the uncertainty relating to model outputs the RP value of the grid cell could belong to the lower neighboring class "Medium". For the EU-15 area, 30% of the grid cells were characterized by a modeled RP in the range of +/−0.20 of the lower or upper limit of the related RP class and therefore could belong to the lower or upper neighboring class.

RP was calculated from the amount of non-irrigated arable land per grid cell because predictions were to be combined with a runoff–response relationship that was established in an agricultural area where this type of arable land is predominant. The effect of considering both non-irrigated and permanently irrigated arable land when calculating RP was judged to be limited at the EU-15 scale, because permanently irrigated arable land is abundant only in Greece, Portugal and Spain (54% of the grid cells per country). In those grid cells, permanently irrigated arable land makes up about 25% of the amount of arable land per cell; hence for these regions the difference between the amount of non-irrigated arable land and the amount of non-irrigated plus permanently irrigated arable land is similar to the range of uncertainty that is assumed relating to each of the input parameters of the RP model. In one sixth of these cells with permanently irrigated arable land (i.e. only 9% of grid cells in Greece, Portugal and Spain) RP values are close enough to the upper limit of the RP class that an increase of arable land would result in a higher RP class. Hence, the effects of omission of permanently irrigated land are restricted to three of the 15 EU countries and in these countries are very local.

5.3. Predicted ecological risk

Predicted ecological risk for small streams in the EU-15 countries is mapped in Fig. 4. A general view of the risk map shows the variability in ecological risk and indicates where runoff inputs into streams draining non-irrigated arable land potentially cause long-term effects on aquatic communities. Ecological risk for streams was very low in the mountainous regions such as the Alps due to a lack of arable land. Ecological risk for streams was low in 34% of the grid cells with arable land (e.g., in the south of Sweden and Finland), because RP of the streams was usually low and the majority of stream sites (≥50%) per grid cell were assumed to have upstream stretches to compensate for effects of runoff inputs through recolonization. However, the RP of about 1/10 of the grid cells with low risk was close to the upper class limit (−3.2 to −3.0). This means that with respect to the uncertainty relating to RP (+/−0.2), the RP class of some of the grid cells could be "Medium" instead of "Low". Ecological risk for streams was medium in 30% of the grid cells with arable land (e.g., in the Po delta and the North German Lowlands). For most of these grid cells, RP was medium (i.e. too low to expect significant effects) and combined with an expected frequency of sites without forested upstream stretches of less than 50%. About 2/5 of these grid cells were characterized by an RP value close to the next lower (−3.0 to −2.8) or upper class limit (−2.2 to −2.0), i.e., ecological risk for these cells could be either low or high instead of medium with respect to the uncertainty relating to RP. Ecological risk for streams was high in 17% of the grid cells with arable land (e.g., in Central France and in the southern parts of Germany). Modeled RP was high enough to expect significant effects, but more than 10% of the stream sites per grid cell were expected to have forested upstream stretches. About 1/4 of these grid cells was characterized by an RP value close to the next lower class limit (−2.0 to −1.8), i.e. ecological risk could be medium instead of high due to the uncertainty relating to RP.

Ecological risk for streams was very high in 19% of the grid cells with arable land (e.g., in large parts of Italy, Spain, and the UK). In these regions, RP was high enough to suggest significant effects on aquatic communities and recolonization pools were expected at less than 10% of the stream sites per grid cell. About 1/5 of these sites were characterized by an RP value close to the next lower class limit (−2.0 to −1.8), i.e. ecological risk could be medium instead of very high due to the uncertainty relating to RP. In summary, the uncertainty relating to modeled RP could be relevant for the RP classification of 30% of the grid cells which in turn could play a role in the classification of 25% of the grid cells in terms of predicted ecological risk for stream sites. Uncertainty in RP would not affect the risk

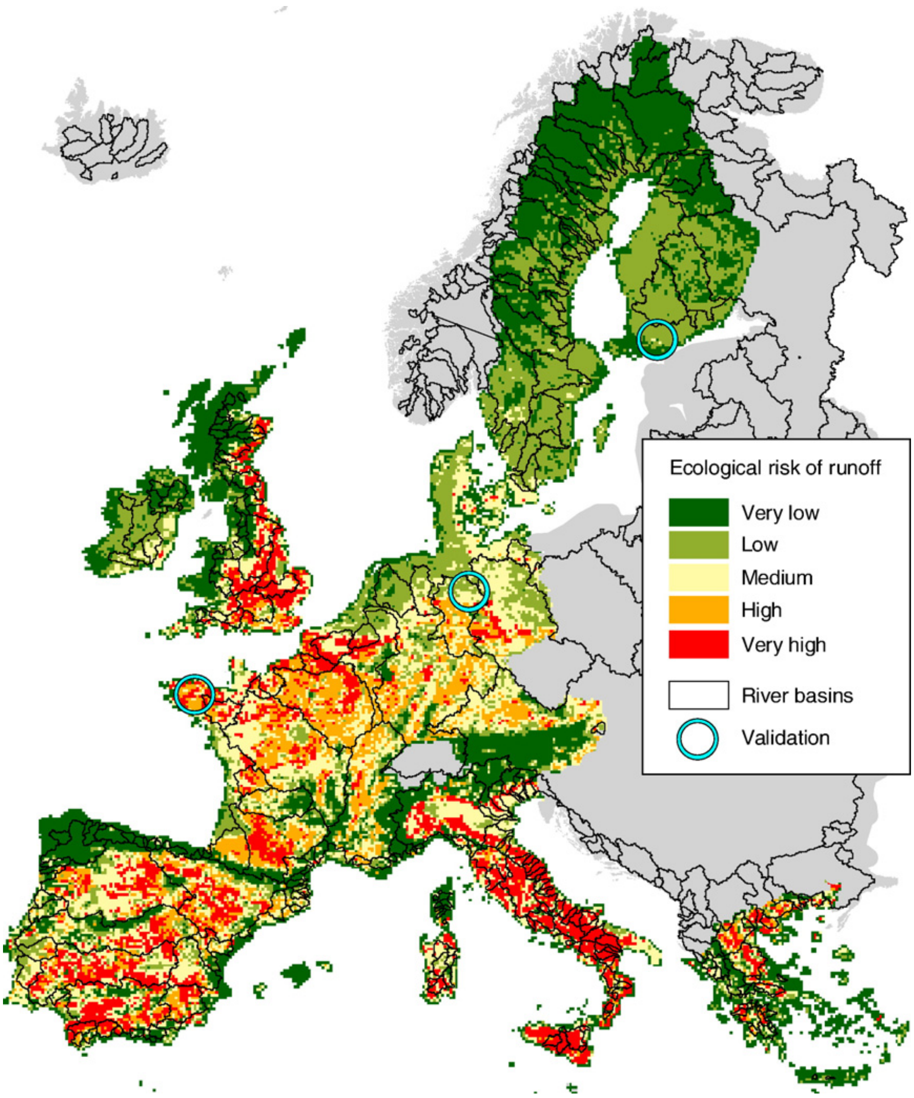

Fig. 4. Distribution of predicted ecological risk of runoff in the EU-15 countries (10-km raster). For details on the risk classification see Table 2.

classification of 5% of the grid cells, because regardless of this uncertainty the RP values of these grid cells would still be classified as high enough to cause significant effects. Uncertainty in RP was not expected to affect the risk classification for the majority of grid cells (75%), which means that the predictions on ecological risk for streams sites within the EU-15 area are robust in terms of uncertainty relating to RP input data.

5.4. Validation

For three selected regions in Finland, France and Germany, predicted ecological risk (i.e., the predicted percentage of stream sites where community composition is affected) was evaluated with the results of field studies of pesticide effects on benthic macroinvertebrate communities. Predicted ecological risk was transformed to the expected number of stream sites per study area where community composition was affected and compared with the observed number of such sites. The 13 Finnish sites are located in grid cells with low predicted ecological risk for streams (Fig. 4). Therefore the predicted median (lower, upper) number of affected stream sites was 1.3 (0; 2.6) and corresponded well to the monitoring results that showed no pesticide effects on the investigated invertebrate communities. The 20 German sites are distributed across grid cells that belong to different classes of predicted ecological risk for streams (Low: 2 sites; Medium: 12 sites; High: 5 sites; Very high: 1 site; Fig. 4). The resulting median (lower, upper) number of affected stream sites was 9.3 (6.1;12.5) and corresponded well to the results of the monitoring study, which showed that community composition was affected at 11 sites. The 16 French sites are located in grid cells with medium (5 sites) or high (11 sites) ecological risk for stream sites (Fig. 4). The corresponding median (lower, upper) estimate of affected stream sites was 9.9 (7.1;12.7) sites and also corresponded well to the number of 9 stream sites that were observed to be affected. In summary, the match of predicted and observed numbers of affected sites suggests that the screening approach presented here produced appropriate estimates of ecological risk resulting from pesticide runoff in the selected regions.

6. Discussion

6.1. Predicting ecological risk at the EU level

The aim of this study was to apply and evaluate a screening model that was designed to characterize ecological risk of pesticide runoff for small agricultural streams at the EU-scale. The main potential of such a screening model is to allow a quick and cost-effective location of areas of concern and thereby to provide information that can help to target regional monitoring programs with respect to the objectives of the WFD more efficiently. Several models have been suggested to identify areas that contribute to diffuse pollution of water bodies from agricultural sources at the EU level. Heathwaite et al. (2003) reported the framework for a simplified screening tool to evaluate the vulnerability of landscapes to phosphorus losses based on the P index. Giupponi and Vladimirova (2006) presented Ag-PIE, a screening model that was applied to assess nitrogen pollution of surface waters and groundwater by chemical fertilizers and manure. For addressing diffuse pesticide pollution at the European level, Tiktak et al. (2004) presented EuroPEARL, a spatially distributed leaching model to predict the contamination of groundwater bodies. Finizio et al. (2005) recommended a procedure to assess mixture toxicity of pesticides in surface water systems at the European level based on standard toxicity data. The approaches mentioned above differ with respect to the pollutants considered, water body types and underlying models. However, they focused on characterization of exposure with reference to one compound only, while none addressed the generic exposure to diffuse pollution due to environmental parameters and none addressed the ecological risk resulting from exposure.

The screening model we suggest differs from those approaches, because it is designed to indicate areas that contribute to diffuse pollution of surface water bodies with pesticides. The model locates areas with a high potential for pesticide runoff into streams and in addition uses a runoff–response relationship to provide an estimate of the related ecological risk.

The scope of our study is to model diffuse pollution in streams and related risk at the landscape level and to validate risk predictions with monitoring data. Essential to this approach is the underlying runoff–response that was established from a large-scale study of effects of agriculture on aquatic invertebrate communities. Another of the few large-scale studies that tried to link exposure and biomonitoring data (Posthuma and De Zwart, 2006) found that predicted effects of observed toxicant mixtures (metals and ammonia or household-product constituents, e.g. triclosan or linear alkylbenzenesulfonate) were confirmed by observed changes in fish species assemblages. The effects were predicted using species sensitivity distributions from laboratory acute aquatic toxicity test data (EC_{50}) and the results showed that in the field there is a relation between exposure to toxicant mixtures and community composition. However, to our knowledge there is no study currently available in which diffuse pollution in streams and related risk (i.e., the expected frequency of affected stream sites per spatial unit) is predicted at the landscape level and in which this kind of risk prediction is validated with monitoring data (i.e., the observed frequency of affected stream sites per spatial unit).

The runoff–response relationship underlying our screening approach was based on the SPEAR concept, which evaluates the sensitivity of communities to pesticides according to the ecological traits of the species that form the communities (Liess and von der Ohe, 2005). Classifying species into SPEAR and species not at risk, the system enables community composition to be simplified and reduces the difficulties of pattern recognition that result from natural variability due to habitat parameters (varying e.g. between eco-regions). Hence the SPEAR concept facilitates distinguishing species variability due to pesticide exposure from variability resulting from differences in other determinants of community composition. The SPEAR concept thus makes it possible to use a runoff–response relationship established for one geographical region to characterize ecological risk of runoff in other European regions, which is supported by the results of the validation presented in this study.

6.2. Robustness of calculated runoff potential

The runoff model underlying RP was developed from a simplified model proposed by the OECD (OECD, 1998). Studies in various geographical regions showed that the OECD proposal for predicting pesticide runoff produced adequate predictions regarding pesticide contamination at different stream sites (Dabrowski et al., 2002; Verro et al., 2002; Berenzen et al., 2005), although the influence of factors such as soil profiles and temperature (Larson et al., 1995; Van der Werf, 1996) is not considered. The runoff model underlying RP describes runoff losses of a compound with generalized properties instead of making predictions for any one substance as is done by the OECD model. The properties of the generalized substance, including the assumption that it does not degrade, represent simplifications that do not apply to most compounds. However, a comparison of predicted RP and measured pesticide runoff for the Braunschweig region showed a good correlation between predicted and observed exposure (Schriever et al., 2007b). Additionally, a comparison of predicted RP and runoff losses, which were calculated for different pesticides (Bach et al., 2000), showed good agreement of designated areas of potentially high runoff for Germany. In summary, modeled RP and the underlying concept of simplified runoff modeling can be considered as suitable for predicting pesticide runoff.

Since modeling RP for the EU-15 area was based on 10-km or 50-km grid information, uncertainties related to the available input data will be more relevant than uncertainties relating to the model structure (e.g. due to buffer strips that are not considered). Particular sources of uncertainty were the resolution of precipitation data and crop statistics (Table 1). Precipitation data were available in 50-km grid resolution, which was lower than the resolution of the other data. Crop statistics refer to administrative units (nomenclature of territorial units for statistics, NUTS III), the size of which differs between EU-15 countries. This may result in considerable spatial variation in the accuracy of data on crop distribution and influences the uncertainty in application rates.

Model sensitivity was assessed by the sensitivity index S (Jorgensen, 1995). Probst et al. (2005a) applied this index to assess the sensitivity of the OECD model when it is used to predict losses from one homogenous patch of arable land. They found that model sensitivity was highest for the parameters buffer width and plant interception. For the current investigation, the sensitivity index was calculated for the model yielding gLOAD, i.e. generalized runoff exposure of a stream site resulting from a heterogeneous set of cultivated fields. The model underlying RP, which did not consider the influence of buffer strips because data were not available, was found to be very sensitive to changes in the parameters plant interception, slope, and precipitation. Uncertainty in RP was not expected to affect the risk classification for the majority of grid cells (75%), which means that the predictions regarding ecological risk for streams sites within the EU-15 area are robust in terms of uncertainty relating to RP input data.

6.3. Considerations on the runoff–response relationship

The relationship between increasing RP and long-term changes in communities was established for streams that were potentially influenced by pesticide runoff as the only stressor (Schriever et al., 2007a). Agriculture, forest, and pasture were the predominant land use in the near-stream environment, while other potential point sources of water pollution did not appear to influence community composition. Therefore, the runoff–response relationship may change in that long-term effects potentially start to emerge at lower levels of RP, if further stressors are present (e.g., runoff from developed land along streams (Lenat and Crawford, 1994) or effluents from waste-water treatment plants (Kosmala et al., 1999)). On the other hand, the runoff–response relationship may change in that long-term effects potentially start to emerge only at higher levels of RP if environmental stressors result in communities characterized by a high recovery potential. For example, drought periods have been reported to favor invertebrate species with short generation times and high dispersal ability (Boulton, 2003). Hence, species with these ecological traits may be dominant in geographical regions where periodic droughts are typical. As a result, communities in these regions may be more robust to environmental stressors including runoff. At sites with a high magnitude of environmental stressors, this would result in a specific relationship between exposure and response that differs from the assumed relationship and may result in discrepancies between actual and predicted ecological risk.

Furthermore, it was assumed that only forested stretches could represent pools for recolonization. While this was demonstrated under Central European conditions (Liess and von der Ohe, 2005; Schriever et al., 2007a), other types of land use that do not involve disturbance by agricultural activities (e.g. extended grassland) may provide additional recolonization pools in other geographical regions. As a consequence the actual ecological risk in these regions would be lower than the ecological risk that would be predicted due to the absence of forested stretches. Regional studies on runoff–response relationships with respect to different types of recolonization pools will help to decrease the uncertainty related to this assumption.

6.4. Use of the screening approach

The screening approach may be applied wherever data are available to specify parameters of the runoff model and to quantify recolonization areas like forest cover with resolution similar to the data we used in this study. The generated maps are relatively easy to interpret and facilitate communicating areas of concern, where the need for site-specific assessment is indicated. Besides identifying areas of concern according to the current situation, this approach could be employed for scenario simulations such as to assess the influence of changes in land use along hydrological networks due to specific strategies of exposure management or due to effects of climate change.

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