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Agricultural drainflow: Landscape risk mapping to facilitate field-level risk assessments of plant protection products in Great Britain

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Abstract

To date, there are very few studies focusing on risk mapping of agricultural drainflow losses of pesticides in Great Britain (GB). A spatially distributed landscape implementation of the U.K. higher-tier drainflow environmental risk assessment modeling framework at 1-km resolution was developed for three sample pesticide active substances and used to derive substance-specific drainflow risk index (DFRI) maps covering GB. The DFRI mapping approach comprised (a) definition of high-resolution landscape scenarios in the drainflow simulation modeling that would underpin the DFRI and (b) construction of two- and three-class bivariate DFRI maps aligned with the existing GB regulatory approach for use in field-scale risk assessment and communication. Visual comparison of the DFRI maps for the three active substances indicates the influence of substance properties, with the mobile nonpersistent substance demonstrating large areas of safe use and with the slightly mobile, moderately persistent substance demonstrating less low-risk areas. Across all three substance DFRI maps, there are locations where no/few drained soils are present, indicating that inherently safe usage locations exist alongside the low-risk areas. A limiting factor of the DFRI approach is the reliance on generalized soil association data as an accurate representation of the soils actually present at a location. The DFRI approach offers an opportunity to move away from a binary regulatory decision of a product being registered or not to one that allows use in locations where it is safe to do so while prohibiting use in those locations where it is not. This more targeted approach would allow users access to a wider range of crop protection options to leverage greater benefits while reducing risks to aquatic habitats. To adopt such an approach, regulators would require reassurance that users could undertake and comply with the results of a field-level risk assessment.

Keywords: drainflow risk index, surface water, pesticides, spatial modeling, risk mitigation

Introduction

Agricultural drainflow is a key transport pathway for pesticides entering surface water bodies in many European countries (Brown & van Beinum, 2009; Kobierska et al., 2020). A large proportion of the agricultural land in the United Kingdom requires subsurface artificial soil drainage systems. The Agriculture and Horticulture Development Board (AHDB, 2015) estimated that around 6.4 million hectares of agricultural land in England and Wales is drained. Anthony et al. (2012b) estimated that the proportion of arable and improved grassland requiring drainage that is currently drained ranges from 46% in Wales to 83% in Scotland. This drainage allows the cultivation of crops on soils, which would otherwise be too wet for economic crop production (AHDB, 2015). The drainage systems artificially speed up the passage of water through the soil profile and into surface waters. This increases the potential for pollution of surface water bodies by pollutants such as pesticides transported from agricultural fields by this drainflow (Health & Safety Executive [HSE], 2025a). Risk mitigation options are very limited for reducing the transfer of pesticides to surface waters in subsurface drains, particularly on heavy clay soils where preferential flow pathways can bypass the bulk soil and provide direct connection between the upper soil layers and the drainage system (Brown & van Beinum, 2009; Hughes et al., 2015; Villamizar et al., 2020).

Currently, risk mitigation measures in the U.K. regulatory risk assessment of pesticides focus on field exposure of surface waters via spray drift, including no-spray buffer zones, use of specific spray techniques, and restrictions in the application window to avoid periods of higher risk to edge-of-field surface

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water bodies via agricultural drainflow (HSE, 2025a). However, research assessing the scope and effectiveness of risk mitigation measures in agricultural landscapes—including consideration of spatial variability, critical source areas, transport pathways, specific vulnerability, and risk mapping using field-scale or landscape models—has gained increasing attention in recent years across Europe as an effective tool in the sustainable management of diffuse pollution from pesticide use in agriculture (Quaglia et al., 2019, 2024; Villamizar et al., 2020; Wendell et al., 2024). Most landscape-specific vulnerability assessment and risk mapping approaches focus on surface runoff losses of pesticides (Hughes & Attwood, 2009; Quaglia et al., 2019, 2024). To date, though, there have been very few studies focused on vulnerability or risk mapping approaches of agricultural drainflow losses of pesticides in the United Kingdom and Europe (Department for Environment, Food and Rural Affairs [Defra], 2011; Hughes et al., 2015; Sandin et al., 2018).

This study outlines a refinement and extrapolation of the existing regulatory higher-tier drainflow (HTDF) environmental risk assessment (ERA) approach for the registration of pesticides in the United Kingdom (HSE, 2025a) to develop a pesticide drainflow risk index (DFRI) mapping approach for possible use in future field-scale risk assessments in Great Britain (GB). The refined approach is deliberately aligned with the existing U.K. HTDF ERA to aid interpretation and adoption of the approach while drawing on high-resolution best available spatial datasets to define detailed landscape scenarios for use of the results by pesticide users within field-scale risk communication and decision support systems. An example is a risk map that would inform users of the locations where it might be safe to use specific plant protection products, accompanied by a decision support tool to inform user behavior to mitigate drainflow risk to surface waters in these and more inherently risky locations.

Such tools not only facilitate the adoption of spatially varying label restrictions for drainflow risk mitigation in the United Kingdom but also have the potential to allow users to make better decisions regarding crop protection management under the existing regulatory regime through, for example, targeting land use or catchment management planning, as well as promoting different user behaviors on higher-risk fields. Targeted field-level risk assessment and management would allow users access to a wider range of crop protection options to leverage greater benefits for themselves and the environment, for instance, by better addressing the long-term risk of pesticide resistance (Shaw et al., 2023) while reducing risks to aquatic habitats.

U.K. regulatory background

The U.K. surface water exposure via drainflow ERA comprises a first-tier drainflow calculation and, if safety is not demonstrated, is followed by an HTDF ERA. The HTDF ERA approach considers modeled daily predicted environmental concentrations (PECs) in a standardized surface water ditch (1 m wide, 0.3 m deep, 100 m long) along the edge of a 1-ha drained arable field for 12 scenarios consisting of four drained soil classes (Denchworth, Hanslope, Brockhurst, and Clifton) and three climates (dry, medium, and wet). Daily predicted losses of pesticides in drainage water are diluted into the ditch by assuming that the drainflow originating from the 1-ha field is instantaneously mixed into the ditch, where the ditch water is refreshed daily. The maximum daily predicted environmental concentrations in surface water (PEC_{SW}) from each of 30 years of simulation are generated and compared with a regulatory acceptable concentration (RAC) to estimate the numbers of "exceedance years" where the maximum daily concentrations exceeded the RAC on at least one day (HSE, 2025a). Two approaches are outlined by HSE for analysis of the modeled daily PEC_{SW} outputs:

Approach 1: Individual exceedance years. Depending on the trophic level determining the RAC, the thresholds used for demonstrating safety vary, with no more than 60% exceedance years (18 out of 30 years) allowed in each scenario for algae and aquatic plants. For fish and aquatic invertebrates, no more than 10% exceedance years (three out of 30 years) are allowed in each scenario.

Approach 2: Weighted level of exceedance. In this approach, which is considered a refinement step from Approach 1 and is always required, the number of exceedance years are calculated per Approach 1. In addition, each scenario is weighted according to the spatial extent of the crop that it represents. The thresholds per Approach 1 apply with a weighted level of exceedance that needs to be $<$10%.

Approach 2 was historically the primary risk assessment approach in the United Kingdom (Brown et al., 2004), with Approach 1 introduced in 2019 in response to the SEISMIC software tool, which was used to define the crop extents (Hallett et al., 1995; Hollis et al., 1995), becoming unavailable for purchase. In June 2025, HSE announced the publication of a SEISMIC Replacement Tool that was developed to facilitate Approach 2 calculations (HSE, 2025b). This study, conducted in the intervening period, utilized an analogous spatial approach for defining the location of and characterizing the landscape scenarios used in the development of the risk mapping approach, ensuring strong alignment with the standard regulatory U.K. HTDF ERA. The current study focuses on GB, as Northern Ireland is subject to U.K. and European Union pesticide registration regulations and the soils dataset for Northern Ireland was not readily available.

Materials and methods

Development of the DFRI approach comprised two primary components: first, definition and use of high-resolution landscape scenarios in the drainflow simulation modeling that would underpin the DFRI; and second, construction of the DFRI aligned with the existing regulatory U.K. HTDF ERA for use in risk communication and field-scale risk assessments.

Simulation modeling

Model

The flow and fluxes of water and pesticide lost to surface water via agricultural drains and the resulting edge of field daily ditch concentrations were modeled in MACRO 4.3 (Jarvis, 1994) in accordance with HSE Chemicals Regulation Division (CRD)

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guidance and modeling tools for HTDF ERA in the United Kingdom. While the most recent HSE guidance and tools utilize MACRO 5.2 (HSE, 2025a), this project utilized MACRO 4.3, as the soil parameterizations for a wide range of drained soils were readily available for this version. The MACRO model was utilized as it is a dual-porosity model that simulates movement of pollutants via chromatographic and preferential flow. The MACRO model is primarily used to predict pesticide leaching through structured soils where macropores play a significant role. Detailed description of the MACRO model and treatment of preferential flow, transport processes, and model settings have been provided in the supplementary materials with sample input parameter files.

Soils

The HSE U.K. HTDF ERA is composed of 12 scenarios that consider four drained soil classes (Denchworth, Hanslope, Clifton, and Brockhurst) to capture the diversity of drained soils within the U.K. agricultural landscape. In previous U.K. policy projects, an expanded landscape modeling soils typology was employed to capture a wide diversity of agricultural soils (see online supplementary material Table S1A and Figure S1), including 12 drained soil classes (e.g., Anthony et al., 2012a; Hughes et al., 2014, 2015). These soil parameterizations have been assessed for suitability in landscape modeling. For example, Hughes et al. (2015) found that catchment total field risk scores correlated well with modeled estimates of risk produced by daily catchment scale models, while Jones et al. (2017) found that simulated catchment pesticide concentrations correlated well with catchment Species at $Risk_{PESTICIDE}$ indicators derived from observed macroinvertebrate assemblages. This drained soils typology (Table 1) was utilized as it offers a more detailed discrimination of drained soils within the United Kingdom as compared with the HSE CRD 4 drained soils class typology. This offers the opportunity for the risk of drained soils to be less constrained by the soil class into which they are categorized and better represented by the representative soil of these classes. It also offers the opportunity to utilize the undrained soils parameterized by this typology in higher-tier refinements considering surface runoff—for example, to address changes in the protection goal that may result from spatially varying label restrictions. These soil classes were based primarily on hydrological response of the soils (see online supplementary material Table S2) with consideration of the soil textural class and distribution of organic matter (see online supplementary material Table S3) within the soil profile. Representative soils for each class were selected on the basis of dominance within and representativeness of the group (Hughes et al., 2015).

Soils parameters required by MACRO were derived according to HSE CRD regulatory guidance (Beulke et al., 2002) and MACRO model guidance (Jarvis, 1994; Jarvis et al., 1997; Stenemo & Jarvis, 2003) using soil property data taken from NATMAP Horizon attribute tables (Fundamentals, Hydraulics, Structure Version 2; Soil Survey Staff, 1983) for the representative soils. Twenty-two soil layers were defined for each representative soil profile in accordance with FOCUS (2015) groundwater guidance. The drain depths and spacing were based on reported values while the bottom boundary condition was set per expert judgment (Table 1). Sample MACRO model parafiles showing input parameters and settings for preferential flow modeling are presented in online supplementary material Table S1B. The spatial extent of each soil class for England and Wales was based on the 1-km resolution "gridded vector" NATMAP product (Cranfield Environment Centre, 2021). The Scottish National Soil map (James Hutton Institute, 2021) was converted to a 1-km

Table 1 Agricultural drain depth and spacing for the 12 drained soil classes.

			Model drain		Data sources[a]		
Drained soil class	Representative soil	Drain type	Depth, m	Spacing, m	Published drain spacing range, m	Published drain depth range, m	Bottom boundary condition
D1	Blackwood	Tile	0.9	30	20–40	0.8–1	Zero flow
D2	Park Gate	Tile	0.9	20	20	>0.9	Zero flow
D3	Wallasea	Tile	0.9	20	20–40	0.75–1	Zero flow
D4	Wigton Moor	Tile	0.9	20	20–40	0.8–1	Zero flow
F1	Hanslope	Mole	0.55	2	40–80[b]	0.75–0.9[b]	Percolation rate: water table height
F2	Bromyard	Tile	0.8	20	10–20	0.75–0.9	Percolation rate: water table height
F3	Whimple	Tile	0.8	20	15–40	0.75–0.9	Percolation rate: water table height
G1	Dale	Tile	0.8	20	15–40	0.75–0.9	Percolation rate: water table height
G2	Dunkeswick	Tile	0.8	20	15–40	0.75–0.9	Percolation rate: water table height
H1	Denbigh	Tile	0.8	20	20–30	0.75–0.9	Zero flow
K1	Denchworth	Mole	0.55	2	20–40[b]	0.75–0.9[b]	Zero flow
K2	Wickham	Tile	0.8	20	15–40	0.75–0.9	Zero flow

[a] Jarvis (1984); Ragg et al. (1984).
[b] Drain depth and spacing do not reflect the additional use of mole drains and subsoiling on these soils.

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"gridded vector" format to match. Soil series in both datasets comprising GB were classed into the 12 drained soil classes.

Cropping

The crop extents (see online supplementary material Figure S2) were defined using the EDINA 5-km resolution agricultural census statistics for 2010. The EDINA (2021) dataset was selected as it is readily available and covers all of GB. The selection of year 2010 was driven by the availability of like-for-like agricultural census classes across GB.

Weather

The HSE HTDF ERA considers three climate zones based on mean annual precipitation (MAP) to define dry (MAP <625 mm), medium (MAP, 625–750 mm), and wet (MAP >750 mm), utilizing the daily weather data from a representative site for each zone. Historical HTDF ERAs utilized daily weather generator data available within SEISMIC (Brown et al., 2004). More recently, the HSE MACRO 5.3–based HTDF ERA (HSE, 2025a) uses updated daily weather (1991–2020) taken from the Monitoring Agricultural ResourceS (MARS) Agri4Cast (2021) gridded agrometeorological data for grid squares coincident with the representative sites for the climate zones. The landscape modeling approach in this study mirrored these developments using daily weather data from MARS Agri4Cast for the 1991–2020 period, with a two-year model warm-up period (1989–1990), for all grid squares across GB. Aside from driving the hydrology in the simulation model, the precipitation data are also used to determine appropriate pesticide application dates using the HSE methodology.

Spatial model

The resolution of the gridded landscape model of 1 km was based primarily on the soils datasets. The 5-km cropping dataset was expressed at 1-km resolution to align with this schema. The centroid of each 1-km grid was assigned to a 25-km resolution MARS Agri4Cast grid. Site attributes were also set by the location of the 1-km grid square simulated (e.g., site latitude).

Substances and application scheme

Three pesticide active substances commonly used in U.K. agriculture with different substance properties were assessed (Table 2), utilizing the registered maximum application rates to winter cereals for products containing these active substances. The application timing of November 8 was based on the AppDate (Klein, 2019) FOCUS D2 scenario for preemergence/early postemergence application with no crop interception as a worst-case assessment. FOCUS surface water defaults were used for all other parameters (see online supplementary material Table S4).

Risk classification

The purpose of the field-level DFRI approach is to inform users of the potential risk to edge-of-field surface water waterbodies receiving drainflow from fields treated with a product containing a specific substance as part of a local risk assessment such that they can make location-specific decisions and (a) not use the product, (b) gather more information to better characterize the field and thereby refine the risk, (c) use the product in a specific way or at a specific time to minimize risk to local water bodies, or (d) use the product without further restrictions according to the label. To align the DFRI with the current HSE scenario-based approach, the number of years exceeding the RAC for each drained soil class was calculated and assigned to a class (as shown in Tables 3 and 4) using the thresholds defined by HSE with and without an uncertainty factor—for example, 18 ± 3 out of 30 years for aquatic macrophyte and algae–based RACs. In addition, two types of DFRI are defined by the data that underpin them: default field attributes based on best available public data (e.g., from digital soil maps) and known field attributes based on knowledge and data held by the landowner/tenant. A three-class risk index approach for field-level risk communication is also shown in Table 5.

Table 2 Chemical input parameters and application rates for the three sample active substances used for the DFRI mapping, as well as the RAC effect thresholds used for risk assessment.

Active substance: Parameters	Value	Application rate, g/ha	RAC, µg/L
A		1 × 60	0.025
Molecular mass, g/mol	394.3		
K_{FOC}, mL/g	3,417		
Freundlich 1/n	0.917		
Soil DT50 (20 °C, pF2), days	143.2		
Plant uptake factor	0		
B			
Molecular weight, g/mol	363.3	1 × 240	0.755
K_{FOC}, mL/g	221.25		
Freundlich 1/n	0.919		
Soil DT50 (20 °C, pF2), days	12.09		
Plant uptake factor	0		
C			
Molecular weight, g/mol	503.5	1 × 7.5	0.129
K_{FOC}, mL/g	64		
Freundlich 1/n	0.91		
Soil DT50 (20 °C, pF2), days	49.72		
Plant uptake factor	0		

Note. DFRI = drainflow risk index; RAC = regulatory acceptable concentration; K_{FOC} = Organic-carbon normalized sorption coefficient; DT50 = Time for 50% dissipation/degradation; pF2 = Soil moisture condition at ∼100 cm water suction (∼10 kPa).

Default field attributes

For use in land use planning and catchment management, the characteristics of fields/locations used to derive the DFRI may be described by default attributes from the best available datasets. In this case, the maximum and minimum soil class exceedance for each 1 km grid is calculated. The location risk is then classed by a bivariate schema based on the minimum and maximum exceedances classed into three ranges (Table 3):

- Low risk: The minimum and maximum exceedance years are below the HSE threshold (18 out of 30 years for aquatic macrophyte and algae–based RACs and three out of 30 years for fish and macroinvertebrates), adjusted for modeling uncertainty (i.e., 15 or two).

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- Medium risk: The minimum and/or the maximum exceedance years values are between 15 and 21 or two and four years out of 30 years.
- High risk: The minimum and maximum exceedance years are above 21 or four out of 30 years.

An alternate approach would be to class the risk by a bivariate schema based on the minimum and maximum exceedances classed into two ranges without factoring in modeling uncertainty (Table 4):

- Low risk: The minimum and maximum exceedance years are below the HSE threshold (18 out of 30 years for aquatic macrophyte and algae–based RACs and three out of 30 years for fish and macroinvertebrates).
- Medium risk: The minimum exceedance years are below 18 or three out of 30 years while the maximum is above the threshold.

Table 3 Three-class bivariate drainflow risk approach considering modeling uncertainty with absolute threshold values.

RAC basis: Minimum exceedance years	Maximum exceedance years		
Aquatic macrophyte and algae	≤15	15–21	>21
≤15	Low	Medium	Medium
15–21		Medium	Medium
>21			High
Fish and macroinvertebrates	≤2	2–4	>4
≤2	Low	Medium	Medium
2–4		Medium	Medium
>4			High

Note. RAC = regulatory acceptable concentration.

Table 4 Two-class bivariate drainflow risk index approach considering the absolute threshold values.

RAC basis: Minimum exceedance years	Maximum exceedance years	
Aquatic macrophyte and algae	≤18	>18
≤18	Low	Medium
>18		High
Fish and macroinvertebrates	≤3	>3
≤3	Low	Medium
>3		High

Note. RAC = regulatory acceptable concentration.

Table 5 Three-class drainflow risk index approach for field-level risk communication.

RAC basis	Exceedance years		
Aquatic macrophyte and algae	≤15	15–21	>21
	Low	Medium	High
Fish and macroinvertebrates	≤2	2–4	>4
	Low	Medium	High

Note. RAC = regulatory acceptable concentration.

- High risk: The minimum and maximum exceedance years are above 18 or three out of 30 years.

Both bivariate class approaches to some extent rely on a location having more than one drained soil class. For a location with a single drained soil, the maximum and minimum exceedance value is the same, resulting in simplification of the risk index that may not be obvious to the user, which is the case in many parts of Scotland given the simplicity of the Scottish soils map.

Known field attributes

For fields where knowledge of the soil and its drainage status are available, the assessment is conducted via the three-class DFRI approach considering modeling uncertainty. Where the field consists of more than one soil class, the results of each soil are spatially weighted and then classed.

Results and discussion

A spatially distributed landscape implementation of the U.K. HTDF ERA modeling framework at 1-km resolution was developed for three sample active substances (Figure 1) and representative HSE climate zones (Figure 2) and used to derive active substance–specific DFRI maps covering GB (Figures 3–5). The influence of the MARS Agri4Cast weather on the exceedance years results is clear, with blocky effects visible in the results, for example, in North-East England (Figures 1 and 3D, Ellipse I). There are locations where there are no/few exceedances in the minimum and maximum number of exceedance year soil classes (Figures 1 and 3D, Ellipse III), which leads to lower risk defined by the two- and three-class bivariate legend DFRIs—for instance, in East Anglia where cereal production is intensive (see online supplementary material Figure S2) and rates of soil drainage are high (see online supplementary material Figure S3).

This, in part, likely reflects the increased number of soil classes simulated, with some not being risky in most scenarios. This also in part reflects the benefit of spatially distributed modeling that utilizes the weather of the location. For example, the representative weather station of the HSE medium and wet climate zones are located at the upper end of the precipitation distribution encountered within these climate zones, representing a realistic worst case for the zones (Figure 2). This assessment of precipitation highlights a potential deficiency that stems from the MARS Agri4Cast gridded meteorological dataset. Many of the low-risk areas, certainly in the East of England where drainage of soils is common, may also be a function of the low MAP (<600 mm; HSE dry climate has a MAP of 532 mm), which in turn leads to low drainflow risk. In addition, all simulations in this study were for dryland cropping, and no irrigation was applied to the simulated crops, although irrigation may be practiced in the drier parts of GB, notably the East of England (U.K. Irrigation Association, 2020; see online supplementary material Figure S4). While for cereals, the use of irrigation is rare as it is seldom economical (El Chami et al., 2015); this may not be the case for some other crops, such as potatoes, sugar beet, and vegetables, and would require inclusion in simulation modeling for higher-

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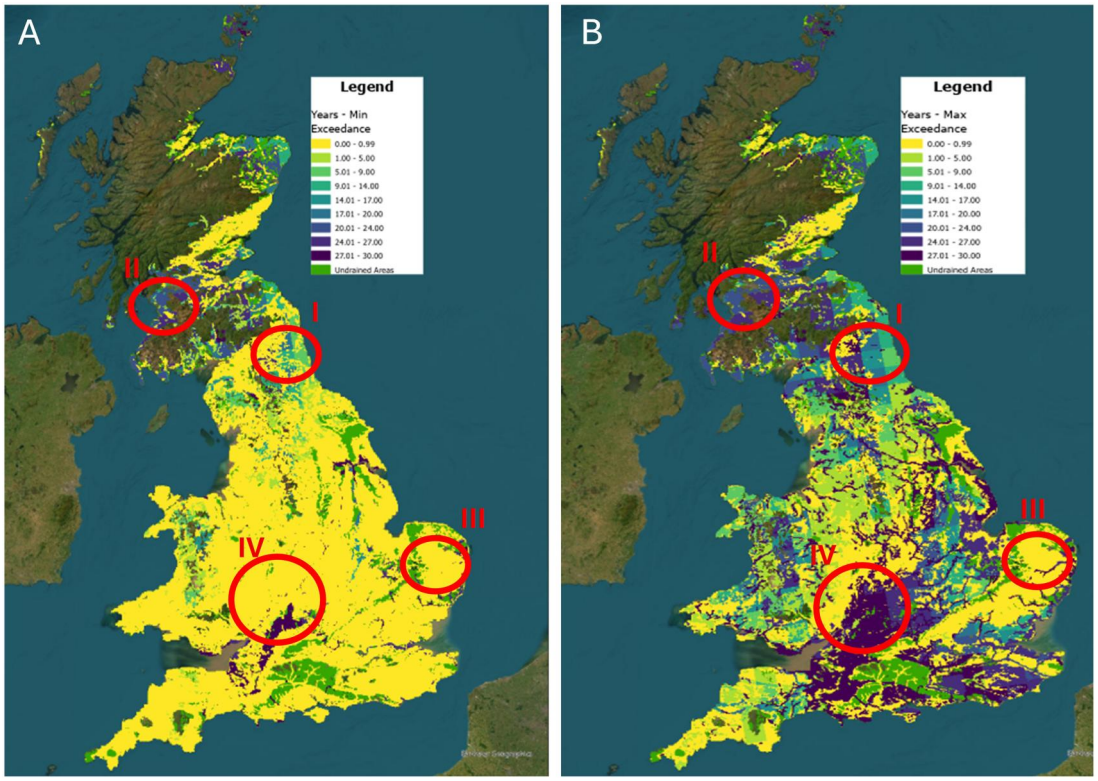

Figure 1 Map illustrating the (A) minimum and (B) maximum number of exceedance years for sample Active Substance A at each 1-km grid location simulated. Red ellipses highlight points discussed in the text relating to the influence of input datasets on the results—namely, (I) MARS weather, (II) Scotland, and (III and IV) England soils datasets. MARS = Monitoring Agricultural ResourceS.

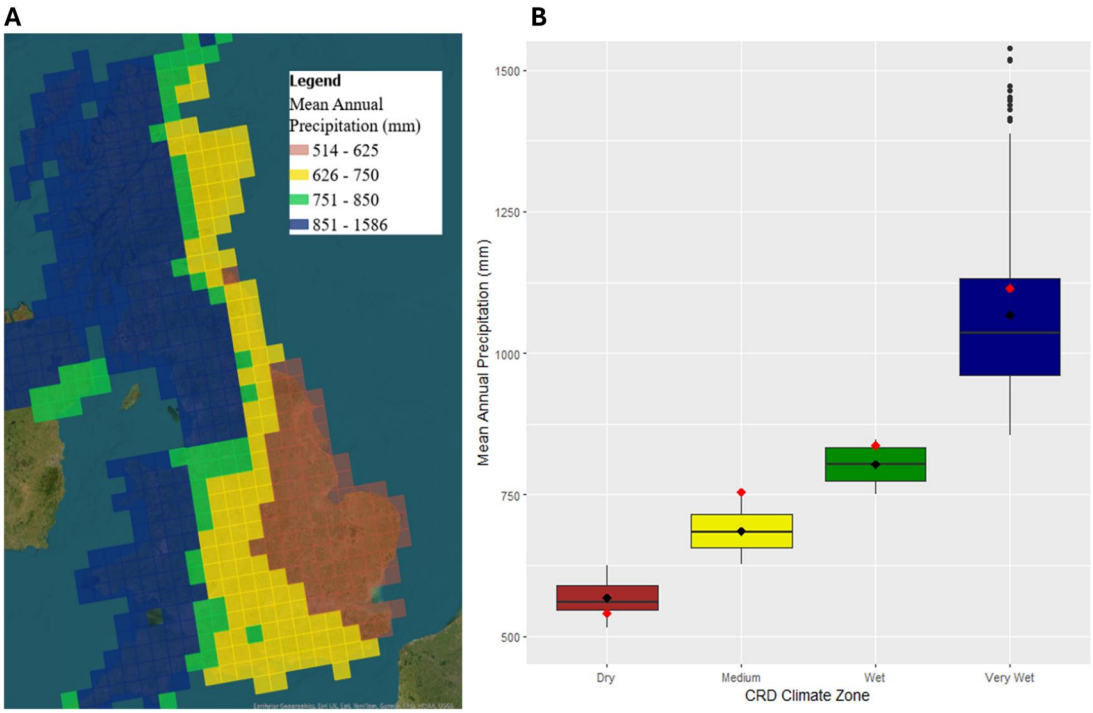

Figure 2 (A) Spatial distribution of mean annual precipitation (millimeters) derived for the 1991–2020 period from the MARS Agri4Cast gridded meteorological dataset. (B) Box-and-whisker plot of the distribution of mean annual precipitation (millimeters) in each HSE climate zone and the location of the representative HSE weather station within this distribution (red diamond). Data are presented as ◆ (black diamond), — (line), □ (box), | (error bars), and ● (circles). MARS = Monitoring Agricultural ResourceS; HSE = Health and Safety Executive; CRD = Chemicals Regulation Division.

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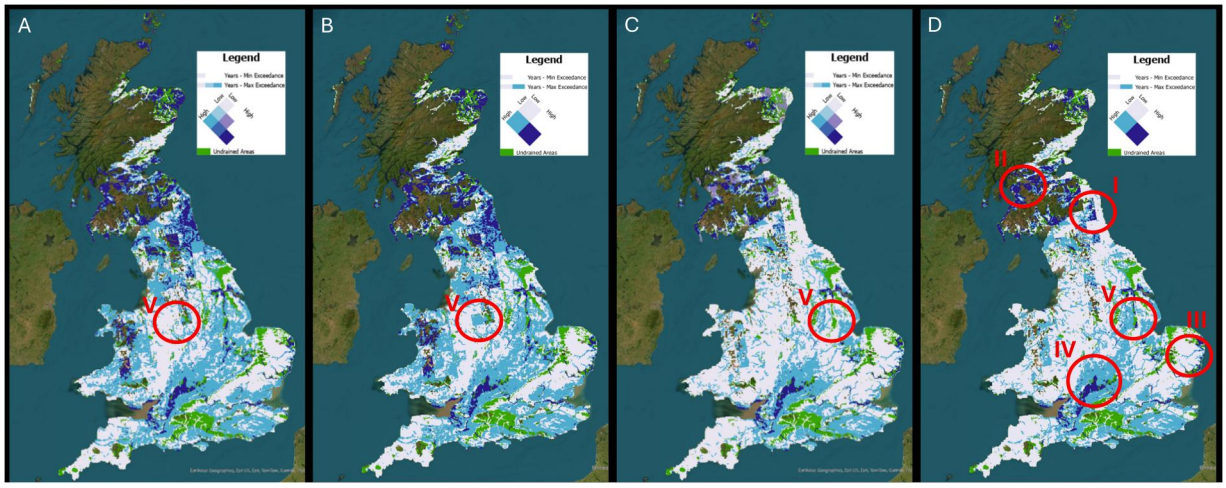

Figure 3 Substance A drainflow risk map illustrates the risk classes defined from the soil classes in each 1-km grid square producing the most and least annual RAC exceedances. Up to three in 30 years may exceed the RAC applicable to fish and aquatic invertebrate following (A) the three-class legend and (B) the two-class legend. Up to 18 in 30 years may exceed the RAC applicable to algae and aquatic plant according to (C) the three-class legend and (D) the two-class legend. The number of years that may exceed the RAC is defined by HSE guidance and relates to the ecotoxicologically relevant species for defining the RAC—specifically, fish and aquatic invertebrate for three in 30 years and 18 in 30 years for algae and aquatic plant. While both RACs are presented here for completeness, usually only the most sensitive species providing the lowest RAC would be used for regulatory risk assessment. Areas in green have predominantly undrained soil classes while those with no risk class have either no arable land or no winter cereals present. Red ellipses highlight points discussed in the text relating to the influence of input datasets on the results—namely, (I) MARS weather, (II) Scotland, and (III and IV) England soils datasets as well as (V) the risk classification approach. RAC = Regulatory Acceptable Concentration; HSE = Health and Safety Executive; MARS = Monitoring Agricultural ResourceS.

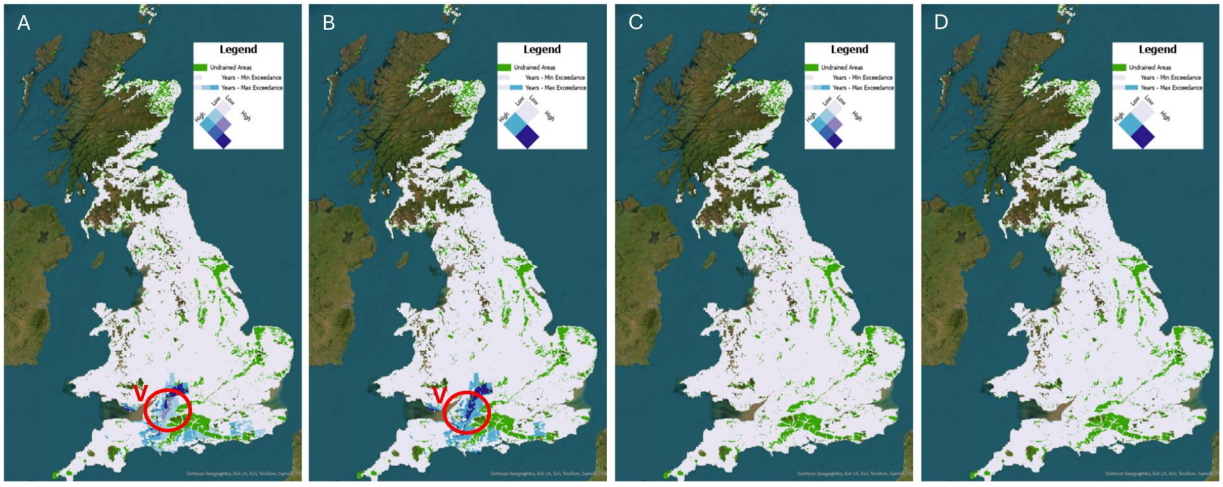

Figure 4 Substance B drainflow risk map illustrates the risk classes defined from the soil classes in each 1-km grid square producing the most and least annual RAC exceedances. Up to three in 30 years may exceed the RAC applicable to fish and aquatic invertebrate following (A) the three-class legend and (B) the two-class legend. Up to 18 in 30 years may exceed the RAC applicable to algae and aquatic plant according to (C) the three-class legend and (D) the two-class legend. The number of years that may exceed the RAC is defined by HSE guidance and relates to the ecotoxicologically relevant species for defining the RAC—specifically, fish and aquatic invertebrate for three in 30 years and 18 in 30 years for algae and aquatic plant. While both RACs are presented here for completeness, usually only the most sensitive species providing the lowest RAC would be used for regulatory risk assessment. Areas in green have predominantly undrained soil while those with no risk class have no arable land or no winter cereals present. Red ellipses highlight points discussed in the text relating to the influence of input datasets on the results—namely, (I) MARS weather, (II) Scotland, and (III and IV) England soils datasets as well as (V) the risk classification approach. RAC = Regulatory Acceptable Concentration; HSE = Health and Safety Executive; MARS = Monitoring Agricultural ResourceS.

value crops where gross margins could potentially support the added costs of irrigation.

In many locations, there is a difference between the minimum and maximum exceedance year results, indicating that in these locations there is a high likelihood that safe usage locations are present (Figures 1 and 3D, Ellipse IV). In contrast, there are locations where the maximum and minimum exceedance year results are high, indicating that great care should be taken when

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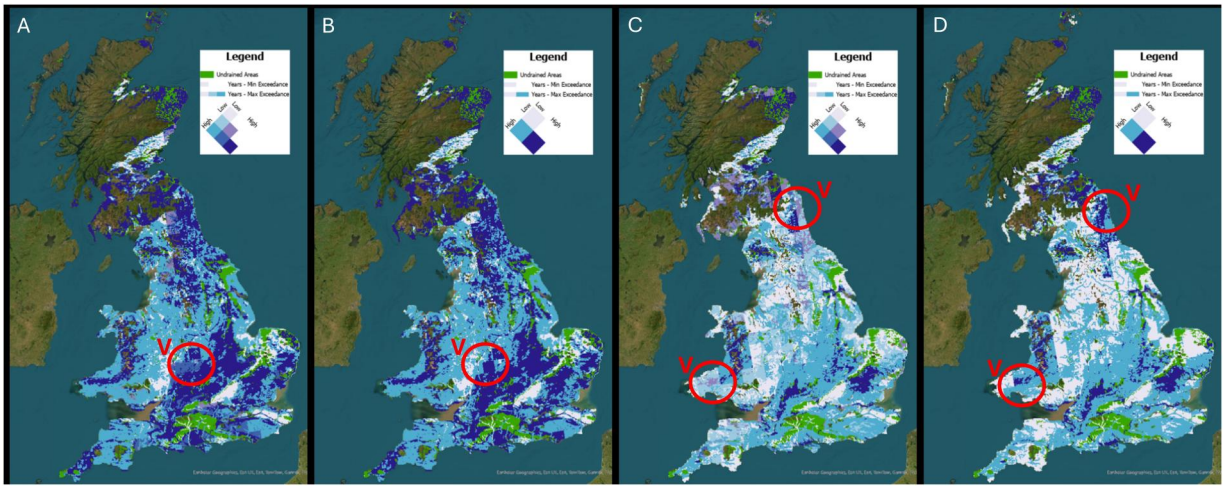

Figure 5 Substance C drainflow risk map illustrates the risk classes defined from the soil classes in each 1-km grid square producing the most and least annual RAC exceedances. Up to three in 30 years may exceed the RAC applicable to fish and aquatic invertebrate following (A) the three-class legend and (B) the two-class legend. Up to 18 in 30 years may exceed the RAC applicable to algae and aquatic plant according to (C) the three-class legend and (D) the two-class legend. The number of years that may exceed the RAC is defined by HSE guidance and relates to the ecotoxicologically relevant species for defining the RAC—specifically, fish and aquatic invertebrate for three in 30 years and 18 in 30 years for algae and aquatic plant. While both RACs are presented here for completeness, usually only the most sensitive species providing the lowest RAC would be used for regulatory risk assessment. Areas in green have predominantly undrained soil while those with no risk class have no arable land or no winter cereals present. Red ellipses highlight points discussed in the text relating to the influence of input datasets on the results—namely, (I) MARS weather, (II) Scotland, and (III and IV) England soils datasets as well as (V) the risk classification approach. RAC = Regulatory Acceptable Concentration; HSE = Health and Safety Executive; MARS = Monitoring Agricultural ResourceS.

using mobile pesticides in these locations as most soils may present a risk to local surface water. In some areas of Scotland, such as Ayrshire, the high minimum and maximum exceedances are a function of the Scottish soil input data, which are less detailed and have only a single drained soil in many locations (Figures 1 and 3D, Ellipse II).

Visual comparison of the two- and three-class DFRI maps suggests that there is little difference between them, yet there are some locations classed as intermediate risk in the three-class system that are pushed into higher-risk categories when the two-class system is applied (Figures 3–5, Ellipses V). Visual comparison of the DFRI maps for the three active substances indicates that substance properties are reflected in the results, with Active Substance B, a mobile nonpersistent compound, demonstrating large areas of safe use, while Active Substance A, a slightly mobile, moderately persistent compound, produces few low-risk locations associated with most locations of intermediate and high risk. Across all active substances as well as DFRI classes, it is important to note that there are locations where no/few drained soils are present, indicating that inherently safe usage locations exist.

As one might expect, the DFRI maps are subject to uncertainty in terms of inputs and model parameterization and design. A representative soil has been utilized for each soil class selected; for example, Denchworth soil is representative of the Denchworth soil class, which is composed of seven soils (after Brown et al., 2004). The degree to which this representative soil represents the simulated drainflow and pesticide leaching behavior of each soil within the class has not been assessed. Similarly, the drain spacing and depth were set on the basis of typical reported values, and drain characteristics in the real world might vary. While the daily MARS Agri4Cast weather data are the best available for spatially distributed modeling, they are interpolated data. The use of the intermediate DFRI that spans the HSE threshold attempts to account for this uncertainty in a simplistic way.

A limiting factor of the default DFRI approach is the reliance on the soils simulated at a location as being an accurate representation of the soils actually present at that location. Anthony (2019) notes that the spatial accuracy of the soil polygons as well as simplification of soil series (Hallett et al., 2017) may lead to significant differences between what the national soils datasets suggest is present at a given location and what soils are actually present at the location (see online supplementary material Table S4). As such, while the default DFRI derived in this study may guide initial user engagement or discussions associated with activities such as regional land use planning or large-scale catchment management, decisions that are related to applied crop protection or that might affect land values should be based on individual field attributes supplied and agreed by the landowner.

Use of the DFRI maps

The DFRI maps offer the opportunity to move away from a binary regulatory decision of a product being registered or not to one that allows pesticide use in locations where it is safe to do so (i.e., locations where edge-of-field waterbody concentrations comply with the RAC) while prohibiting use in those where it is not. The HSE is likely to have two key concerns with spatially varying pesticide label restrictions: ability of the user to comply with the label and changes in the protection goal.

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User compliance

These DFRI maps facilitate engagement with pesticide users whereby recommendations could look akin to the following:

- Undrained (green areas): Confirm soil drainage status (i.e., undrained) and soil texture class (i.e., is an undrained soil series) to use product.
- Low risk (beige areas): Confirm that soil texture class is within the modeled "safe" classes to use product. Existing soil information would be sufficient to confirm the soil texture class.
- Intermediate risk (blue areas): Check soil texture class and drainage status to use product, that is, use the product only if the soil is undrained or a low-risk soil texture class (possibly because actual soils present are different from the default soils suggested and of low risk). This would require detailed knowledge of the soils present (e.g., soil series–level soil identification).
- High risk (dark blue areas): Do not use product. Confirm soil texture class and drainage status, that is, use the product only if the soil is undrained or a low-risk soil texture class. This would require a detailed site investigation conducted by specialists, such as the field-level soil mapping conducted for precision agriculture.
- If the soil is deemed to be undrained for any of the three risk classes, this should be on the basis of soil texture class and a site investigation to confirm the absence of drains according to AHDB (2015) guidance on field drainage.

Given the uncertainties in the soil mapping that underpins the risk maps, while these maps may provide a vehicle for user engagement, it would be wise for all users to conduct a drainflow risk assessment specific for their field where they intend to use the pesticide. In the example provided here, data required to "override" the default increase with the level of risk. The plausibility of such an approach would need to be tested with farmer focus groups, such as those within the eight Catchment Sensitive Farming catchments. This drainflow risk map user engagement should be administered through a web-based portal (see example in Defra, 2011) and the results stored for farm audit compliance purposes (e.g., by the Rural Payments Agency or farm assurance schemes) and for pollution incident investigation by the environment agencies.

The HSE would require reassurance that farmers could comply with a mandatory label restriction that requires them to undertake a field-level risk assessment. A user would need to conduct and document this field-level risk assessment by using an online decision support tool or one built into farm management software, for example. The decision support tool would not only ensure that the assessment was conducted appropriately but would also document the ERA as part of farm payment or assurance scheme inspections. This ERA would require the farmer to provide information for that field, including drainage status, drainage characteristics, and soil characteristics.

Drainage status

The majority of farmers know the drainage status of their land, with only 2% ± 1% to 7% ± 2% of holdings not knowing this status according to two surveys—namely, the ADAS Farmers' Voice and Defra Farm Practices Survey, respectively (Hughes & Attwood, 2009). In the best-case scenario, the farmer has a drainage installation map. This map would show the layout and extent of any piped field under drainage and indicate if further treatments were required, such as mole drainage. In the worst-case scenario, the farmer has no idea about the field drainage status and could either (a) utilize a risk envelope approach and assume the worst-case soil and drainage status (i.e., mole drains) or (b) gather the required information.

Guidance is available (AHDB, 2015) on how to determine the presence and condition of existing field drains. This process would typically involve a field walkover looking for any signs of drainage infrastructure, such as inspection chambers and outfalls. A detailed soil survey could also be used to establish the soil drainage class, and this would indicate the likelihood that land drainage had been installed. When coupled with local knowledge and published drainage extents (Anthony et al., 2012b), the likely drainage status would be clear; for example, an agricultural field located in the Cambridgeshire fens would almost certainly be drained. If farmers wanted to challenge this baseline assessment, they would need to conduct a soils survey and/or intrusive existing drain survey with a mechanical excavator to demonstrate that drains are either not present or not functioning.

There are precedents for approved infield soil drainage assessment methods to obtain regulatory consents. These include soil percolation tests for design of surface water soakaways or treated effluent drainage fields and soil profile examinations to determine land quality. Both of these have approved mandated methodologies to be used by local authorities as integral parts of the planning process (Agricultural Land Classification, 2012; Ministry of Housing, Communities and Local Government, 2010, 2012).

Soil texture

The majority of farmers know the dominant soils on their holding: 84% ± 2% know the dominant soil type for >75% of their fields with 55% ± 2% knowing this for all fields; a combined total of ∼12% of fields have an unknown soil texture (Hughes & Attwood, 2009). The soil series or detailed knowledge of the soil texture is required to class the field soil into one of the model soil classes utilized within the field-level ERA decision support tool. While most commercial farmers would have some basic knowledge of their soil to comply with existing regulations in relation to soil protection and nutrient management, it is unlikely that this information would be definitive for these purposes as such information and assessments tend to focus on the topsoil rather than the full soil profile to below drain depth. In the best-case scenario, the farmer has a detailed field soil map, for example, produced to facilitate precision agriculture, a farm waste management plan, or an Agricultural Land Classification assessment. In a worst case, the farmer would need to have the soil texture of each field assessed and classed into the appropriate model soil class, akin to that undertaken for the field risk tool developed for UKWIR (Hughes et al., 2015). This could be done by either a desk-based crosswalking of an existing assessment with sufficient information or a bespoke field survey, likely part of a field drainage status survey. As with other regulatory soil assessments, these would need to be conducted by a suitably

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qualified and experienced person. Such field soil and drainage assessments would range in cost from approximately £1K to £4K per field depending on survey complexity, field size, and number.

About 6.4 million hectares of agricultural land in England and Wales are drained with piped (tile) drainage systems (AHDB, 2015), which covers about half of the total farmed land in the region. There are no readily available statistics for the overall proportion of mole drains in England and Wales. Anthony et al. (2012c) synthesized all available data on land drainage and concluded for England that 43% of the drainage area was installed prior to 1939 and 38% since 1970. Note that these field-level ERAs would include a number of worst-case assumptions within drainage design—for example, that mole and tile drains are operational across the whole field and that they are operating as efficiently as newly installed drains. Similarly, the field-level ERA considers the worst-case weather years, as these drive the precautionary risk assessment. These factors provide a good margin of protection to account for uncertainties that may arise as part of the field-level ERA using model soil classes informed by suitable field knowledge. The proposed field-level ERA should be considered alongside the finding that a majority of farmers would feel reasonably confident in interpreting a spatially variable pesticide label restriction, with only 12% ± 1% being unsure or not confident to comply with a label restriction prohibiting application to a certain soil type (Hughes & Attwood, 2009).

Protection goal

The U.K. surface water regulatory risk assessment framework does not explicitly consider surface runoff. Instead, the drainflow ERA is considered to define the outer risk envelope, which is also protective of risks from surface runoff, although it should be noted that topographic factors such as slope are not considered within the MACRO model. By labeling off pedoclimatic combinations that present drainflow risk to surface water, the assumption that the drainflow ERA is fully protective of surface water may no longer be valid. This could be addressed in two ways.

First, the R scenario results from FOCUS surface water, which consider topographic factors such as slope, are evaluated for scenarios that are relevant to the United Kingdom and the crop of interest. A previous HSE CRD project (Defra, 2006) has already defined which runoff or R scenarios are relevant to the United Kingdom for different crops (see online supplementary material Table S6). For winter cereals, the R1 and R3 scenarios would be considered. HSE previously developed a U.K.-specific SWASH version, which uses modified versions of the FOCUS surface water scenarios (Defra, 2007).

Second, runoff scenarios for each drained soil class are developed that allow a runoff risk assessment to be conducted specific to the proposed label restrictions. The soil parameterizations for runoff assessments for each soil class simulated have been developed for previous projects and are readily available. The HSE also recently confirmed that it intends to develop runoff scenarios (M. Morris, HSE CRD, personal communication, 2024) to facilitate regulatory refinements, such as spatially varying label restrictions.

These could be built into the same web tool that facilitates the DFRI assessment such that runoff risk is considered alongside the drainflow risk.

Future prospects

The results presented in this article are for three sample substances to demonstrate the utility of the proposed DFRI approach and are not intended to be generalized to other substances but rather serving as a proof of concept. The development of DFRI maps that facilitate engagement with and inform pesticide user behavior at the field level are consistent with the HSE regulatory U.K. HTDF ERA to facilitate ease of regulatory adoption and can address the key concerns of the regulators. While development of DFRI maps for each registered pesticide active substance and uses that would likely utilize spatially varying labels may require significant computational effort, this may be required to ensure environmental safety and promote the adoption of novel field-level ERA approaches. Next steps would be to test the approach with users, for example, in one of the eight Catchment Sensitive Farming catchments. Farmers within these catchments are very familiar with engagement activities, and the catchment officers to undertake the user engagement programs are in place. In addition, the Environment Agency conducts enhanced chemical water quality monitoring for these catchments, which, when supplemented with edge-of-field waterbody monitoring, would allow the impact of adopting spatially varying label restrictions on water quality and surface water risk to be evaluated.

Supplementary material

Supplementary material is available online at *Integrated Environmental Assessment and Management*.

Data availability

Data are available in the Supporting Information. Additional data were made available to the editor and reviewers, enabling thorough assessment during the manuscript's peer-review process.

The soils data—the spatial data that allow them to be plotted and the attribute data that are used to parameterize the soil profiles—are third-party intellectual property and cannot be shared under the terms of the data license agreement. While the soils data license agreement also claims intellectual property ownership over all derived products, we have provided sample model input files for the soil profiles simulated for use by the reviewers should they want to do so. These are a synthesis of the original data and suitably derived such that this is less likely to be an issue for the data ownership but allows us to meet the request for transparency. We have also provided weather files and associated output files for the model simulations, including an example of the results extracted from these.

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Author contributions

Gregory Owen Hughes (Conceptualization, Data curation, Formal analysis, Funding acquisition, Investigation, Methodology, Project administration, Supervision, Validation, Visualization, Writing—original draft, Writing—review & editing), Abdul Abu (Formal analysis, Funding acquisition, Investigation, Methodology, Project administration, Visualization, Writing—original draft, Writing—review & editing), Robin Sur (Conceptualization, Funding acquisition, Supervision, Visualization, Writing—original draft, Writing—review & editing), Adrian Terry (Funding acquisition, Supervision, Visualization, Writing—original draft, Writing—review & editing), and Kirk Hill (Validation, Writing—original draft, Writing—review & editing)

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Conflicts of interest

The authors have no conflict of interest to disclose.

Ethical statements

All authors listed have approved the manuscript and agree to its submission to *Integrated Environmental Assessment and Management*.

The authors also confirm that there are no animals or human subjects involved in this work.

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