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Quantifying the influence of climate change on pesticide risks in drinking water

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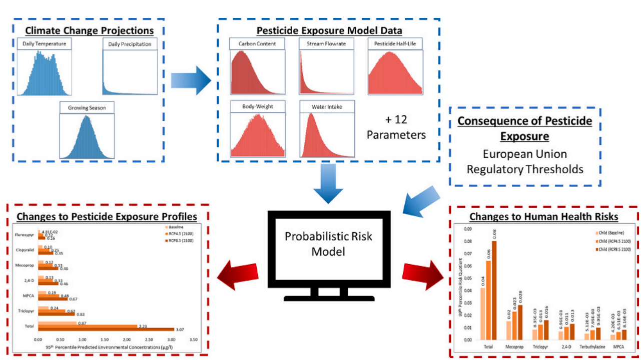
HIGHLIGHTS

- Climate change is likely to impact pesticide contamination of water supplies.
- A probabilistic model to assess future pesticide human health risks was developed.
- It was shown that climate change increases pesticide exposure and risks in Ireland.
- Regional disparities in results emphasise a need for localised climate strategies.
- Proposed approach may be used for public health and climate adaptation strategies.

GRAPHICAL ABSTRACT



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ABSTRACT

Climate change can influence pesticide contamination and resulting human health risks due by altering weather conditions that drive pesticide fate and transport. However limited research has examined these effects, leaving regulatory frameworks and adaptation strategies unable to address future pesticide risks. This study develops a novel probabilistic model to quantify climate change impacts on pesticide-related human health risks under two different climate scenarios, using study locations in the north-east and south-west of Ireland. Results indicate that pesticide concentrations in drinking water are projected to exceed legal limits more frequently, and by greater amounts, under all climate scenarios, with associated health risks increasing by an average of 18 % under RCP 4.5 (2050) and 38 % under RCP8.5 (2100). The model results also indicate significant regional variation in health risk, with risk 48 % higher in the south-west than the north-east under baseline conditions. Climate

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change effects intensify these regional variances with risk up to 70 % higher under RCP4.5 (2050), and 85 % higher under RCP8.5 (2100). Despite these increases, overall pesticide human health risks are likely to remain low in Ireland under future climates. This study presents a probabilistic framework that may be applied internationally to quantify the impact of climate change on human health risk at a local-scale and may be adapted for different site conditions and climate projections to suit users' needs. This approach can inform future pesticide management programmes by identifying vulnerable areas and key pesticides under changing climate conditions, emphasizing the importance of incorporating climate change into pesticide risk mitigation and public health strategies.

1. Introduction

Climate change has emerged as one of the most pressing challenges of the 21st century and has already resulted in extensive changes to the atmosphere, natural resources, and aquatic and terrestrial ecosystems (IPCC, 2022). There is significant concern regarding the impacts that climate change-related flooding and droughts may have on the availability and quality of drinking water resources (Welch et al., 2021; Gascuel-Odoux et al., 2023). In addition to these hydrologic effects, climate change also has great potential to affect drinking water quality due to increased agricultural pollutant loading, including pesticide contamination (Kaka et al., 2021). Shifts in precipitation patterns due to climate change are likely to affect the rates of pesticide runoff to waterbodies, as it was found that rainfall is a key driver of pesticide loss (de Souza et al., 2020). Therefore, climate change has the potential to significantly affect human exposure to pesticides via drinking water supplies due to these changes in contamination rates (Zhu et al., 2021). The extensive use of pesticides in modern agriculture has already resulted in the accumulation of contaminants in soil and water bodies (Li and Niu, 2021; Wolfram et al., 2021; McGinley et al., 2023), posing significant threats to human health and the environment (Kalyabina et al., 2021; Wang et al., 2022). The intensification of food production required to meet a 60 % increase in food demands by 2050 (UNEP, 2022), as well as the projected changes to climate variables (O'Brien et al., 2024), have the potential to exacerbate the likelihood of exposure to these chemicals in drinking water supplies and, in turn, pose greater health risks (Delcour et al., 2015; Saha et al., 2019). Therefore, the implementation of pesticide risk assessment and management strategies are key factors in reducing the future impact of climate change on water quality (Schäfer et al., 2019).

Pesticide risk assessment and management often relies on simulation models to assess pesticide transport and resulting environmental and human (Welch et al., 2021) health risks under current conditions, due to their effectiveness in representing the relationships between pesticide properties and site conditions (Li and Niu, 2021). Several studies have applied deterministic modelling approaches, models that produce a single outcome based on the assumption of fixed relationships between input variables and excluding any randomness, to predict pesticide concentrations for research and regulation under current climatic conditions (Luo et al., 2011; Pullan et al., 2016; Wang et al., 2019). Probabilistic modelling approaches, models that incorporate variability and uncertainty through the use of probability distributions to represent likelihood of different variable combinations and resulting outcomes, are becoming more widely used in this field to account for the variability associated with pesticide behaviour, as well as the uncertainty associated with modelling soil, weather and pesticide data (Cantoni et al., 2021; Troldborg et al., 2022). These deterministic and probabilistic simulation models are combined with pesticide health risk data in order to quantify the risk arising from pesticide usage under current climatic conditions (Focks et al., 2014). However, it has been suggested that environment and human health risks arising from pesticide exposure are likely to change in the future due to climate change (Nadal et al., 2015; Tudi et al., 2021). Many studies have identified parameters which may change under future climates and therefore may influence future pesticide risks i.e. changes to rainfall frequency and intensity, increasing temperatures and changes to growing seasons (Noyes et al., 2009; Delcour et al., 2015; Biswas et al., 2018). Although there is considerable research identifying key climate change parameters, there is still limited work modifying fate and risk models to quantify the influence of these effects on pesticide risk to the environment and human health arising from pesticide use (Hader et al., 2022; Bolan et al., 2024). To date, the majority of pesticide fate and risk models only provide estimation of risk under current conditions, and do not provide insight into how rates of pesticide exposure and health risks may decrease/increase with changes in climatic conditions (Sperotto et al., 2017; Oldenkamp et al., 2024). These approaches cannot provide decision-makers and stakeholders with data needed for future-proofing pesticide regulations and water treatment infrastructure. There is thus considerable need for the analysis of climate change impacts on future pesticide-related health risk to provide invaluable risk-based decision support (Stewart and Deng, 2015).

Although it is still a limited area of research, some studies have begun to integrate climate projections and pesticide fate/risk models (Moe et al., 2024). Initially, work on applying probabilistic approaches to quantify climate change impacts on pesticide loss was limited to assessing changes in future environmental pesticide concentrations in groundwaters (Steffens et al., 2015) and surface water (Gagnon et al., 2016). However, these studies can only be used to assess pesticide concentrations and provide no information about future pesticide risks to human health. There is a pressing need for such research as it provides crucial risk-based information for future-proofing pesticide remediation measures and public health strategies, decisions for water infrastructure investment, and informing adaptive practices to ensure pesticide regulation remains effective in the future (Anik et al., 2023; Bolan et al., 2024). More recently, this work has been advanced by researchers to analyse potential changes to pesticide environmental risks arising from future climate conditions using Bayesian Networks (Mentzel et al., 2022; Martínez-Megías et al., 2023; Oldenkamp et al., 2024). Both Mentzel et al. (2022) and Oldenkamp et al. (2024) adapted a pesticide exposure model (World Integrated System for Pesticide Exposure, WISPE) using future climate projections for Norway to assess the influence climate change will have on pesticide risks aquatic organisms. Similarly, Martínez-Megías et al. (2023) applied Bayesian networks to another pesticide exposure model (RICEWQ) to probabilistically assess environmental risks to aquatic invertebrates and non-target plants arising from pesticide use in rice fields, and how these risks may respond to a changing climate in a Mediterranean context. Although these studies have contributed significantly to the literature by advancing the work in the combination of global climate models with pesticide fate and environmental risk models, these studies do not consider the effects on pesticide-related human health effects. Furthermore, they have been limited due to the use of ensembles with a small number of climate models, low-resolution climate models or the use of projections that have not been bias-corrected. This can introduce model variability and errors into the projections and may not fully capture the spatial variability of climate change which can impact reliability of projections (Navarro-Racines et al., 2020; Kotamarthi et al., 2021; Moe et al., 2022). Despite the work carried out in these studies, research on adapting pesticide environmental risk assessments for climate change effects is still an emerging area of research (Martínez-Megías et al., 2023), and even less work has been carried out in adapting human risk assessments for climate change, with no published work to-date quantifying the potential climate change impacts on pesticide-related human health risks (Gatto et al., 2016; Langreiter et al., 2023). This current study seeks to address the pressing need for pesticide-related human health risk information for future climatic conditions, utilising state-of-the-art climate projections from high-resolution, bias-corrected climate models.

The aim of this study was to develop a novel model framework, incorporating state-of-the-art projections into a probabilistic pesticide health risk model, to predict how future climatic conditions may influence pesticide concentrations in drinking water due to changes in pesticide runoff and assess resulting human health risk for future climatic conditions. This study presents a first step towards quantifying climate change impacts on future risks posed to human health from pesticide exposure using probabilistic modelling. This was achieved by (1) producing probability distributions for climate inputs using multi-model, bias-corrected and detrended projections based on downscaled regional climate models, (2) adapting probabilistic pesticide risk model using these developed climate parameters to quantify changes to pesticide-related human health risks in future climates, (3) illustrating the application of this model to predict future pesticide concentrations and resulting risk under future climatic conditions for Ireland and (4) assessing the influence spatial variability in climate change may have on regional discrepancy in pesticide health risks. This study is one of first to assess the interaction between climate change and pesticide transport in terms of human health risks, and the approach developed may be used to inform pesticide-specific risk management schemes and climate adaptation strategies for the future.

2. Methodology

2.1. Climate change projections

Previous work found that precipitation and growing season length are key climate parameters in pesticide runoff modelling (Harmon O'Driscoll et al., 2024). As growing season length can be determined using a thermal growing season metric dictated by daily temperature (Dunn et al., 2020), temperature is also an important variable to consider. As a result, precipitation, temperature, and growing season length were selected as the primary climate variables for this analysis. In this study standardised, high-resolution climate projections of daily precipitation and temperature data in Ireland were obtained from the TRANSLATE project (O'Brien and Nolan, 2023; O'Brien et al., 2024), however the methodology developed in Section 2.2 may be applied to projections from any regional or global climate model. The TRANSLATE project has developed bias-corrected, detrended and downscaled projections based on post-processing the output of two multi-model ensembles (with overall totals of 27–35 ensemble members, depending on the different future forcing scenarios considered) from regional climate models (COSMO-CLM and Weather Research and Forecasting (WRF) (Flanagan and Nolan, 2020; Nolan and Flanagan, 2020). Projections were obtained for 30-year time periods using different emission scenarios for the quantitative assessment of climate change impacts within Ireland. Location-specific climate projections for two Irish sites in the north-east (NE) (Dunleer, Co. Louth; 53° 50′ 09″ N, 6° 25′ 04″ W), and south-west (SW) (Causeway, Co. Kerry; 52° 24′ 48″ N, 9° 43′ 50″ W), were used in pesticide risk modelling to represent the regional variability of climate change effects (Fig. S1 in the Supplementary Information). Both the NE site and the SW site have relatively similar site conditions, i.e. land-use and soil types, as they are predominately grassland (pasture, rough grazing, silage, hay or any combination) (EEA, 2021) with similar soil types (both sites classified as hydrologic group C soils: loamy soils with moderately high runoff potential (USDA, 2009; EPA, 2021). These site conditions are representative of the average Irish agricultural scenario, where 90 % of agricultural land is grassland, and group C is the dominant soil group (Cawkwell et al., 2017; EPA, 2021). Five discrete warming scenarios were considered in the analysis: (1) baseline existing conditions, based on 1976–2005 climatic data, (2) the IPCC RCP4.5, intermediate emissions scenario, for mid-century (2050), (3) the IPCC RCP8.5, high emissions scenario, for mid-century (2050), (4) RCP4.5 for end-of-century (2100), and (5) RCP8.5 for end-of-century (2100). The two timeframes (mid-century and end-of-century) are based on 30 years of climate projections over the period of 2041–2070 and 2071–2100, respectively. The changes to the average precipitation and temperature based on these projections are given in Table 1, where the values represent the change under each climate scenario relative to the baseline conditions.

Climate projections for Ireland were integrated into the probabilistic risk model framework (Section 2.2) to illustrate how the proposed framework can be applied, using data from climate models for specific regions. However, other countries across the North Atlantic region (Northern Europe, Eastern Canada and North-Eastern USA) are likely to experience relatively similar climate effects to Ireland. IPCC projections for the region suggest that precipitation will experience a median increase of 12.2 % and temperature by a median increase of 2.1 °C by 2100 relative to 1986–2005 levels (Gutiérrez et al., 2021). Precipitation and temperature in Ireland are projected to increase by 10.8–12.2 % and 2.2–2.4 °C, respectively (Table 1), which is in line with increases across the North Atlantic region. Therefore, while the projections used in this study are specific to two locations in Ireland, the results in this study may be indicative of changes to future pesticide risks across the region and may help inform the need for pesticide risk assessment under future climates.

2.2. Pesticide risk assessment methodology

This work builds on the authors' previous work (Harmon O'Driscoll et al., 2024), through re-development of a user-friendly, probabilistic pesticide risk model to consider the influence of climate change using state-of-the-art climate projections. These modifications enabled the quantification of climate change impacts on future pesticide runoff to surface waters and resulting human health risks arising from changes to pesticide exposure via drinking water. In brief, the authors' previous work developed a probabilistic pesticide risk assessment by building on the existing Simplified Formula for Indirect Loading caused by runoff (SFIL) (OECD, 2000; Berenzen et al., 2005). This allowed users to assess the influence of land use, soil and climate conditions, site data, and pesticide properties on exposure concentrations of pesticides in surface waters. The authors modified this base model by (1) incorporating a more detailed approach to assess the rainfall-runoff relationship using the Soil Conservation Service (SCS) curve number method (USDA, 2004b), (2) including additional in-stream processes that may affect pesticide concentration, (3) combining this pesticide transport model with a widely-used pesticide health risk assessment approach (FAO and WHO, 1997; EFSA, 2019), and (4) utilising a Monte Carlo simulation

Table 1
Changes to parameters under future scenarios relative to baseline (where μ = mean; σ = standard deviation).

Scenario	Daily precipitation		Daily mean temperature	
	Change μ (%)	Change σ (%)	Change μ (%)	Change σ (%)
Dunleer (North-East Site)				
RCP4.5 (2050)	+ 3.1	+ 7.1	+ 10.2	+ 3.9
RCP8.5 (2050)	+ 5.4	+ 10.4	+ 14.7	+ 4.8
RCP4.5 (2100)	+ 6.8	+ 11.5	+ 14.4	+ 3.9
RCP8.5 (2100)	+ 12.2	+ 19.9	+ 26.9	+ 6.6
Causeway (South-West Site)				
RCP4.5 (2050)	+ 2.0	+ 13.4	+ 8.7	- 1.0
RCP8.5 (2050)	+ 5.1	+ 19.6	+ 12.4	- 0.1
RCP4.5 (2100)	+ 5.8	+ 19.7	+ 12.6	- 0.7
RCP8.5 (2100)	+ 10.8	+ 21.7	+ 22.9	+ 1.2

approach for probabilistic assessment. The SFIL base-model has been widely used to assess pesticide runoff concentrations and has been found to perform relatively well compared to monitored pesticide concentrations in several regions globally (Schriever and Liess, 2007; Isaltino et al., 2015; Utami et al., 2020). This modelling approach could potential also be applied to predict runoff of other organic chemical pollutants to suit users' needs, as it has been suggested that the processes of chemical transport and degradation is common to the majority of organic chemicals (Di Guardo et al., 2018). However, this approach may not account for key parameters or processes specific to different pollutants, for example conventional fate models are not suitable for PFAs or polar substances such as PCBs due to data gaps or non-standard fate processes (Schlüter et al., 2022). Therefore, these factors need to be considered should this approach be applied to other chemicals beyond pesticides.

For the work detailed in the current paper, the model was adapted to facilitate incorporation of climate projections from a range of downscaled regional climate models to account for the direct effects of climate change on pesticide transport and resulting health risks, thereby illustrating how the model may be adapted for future climates to inform climate adaption strategies. Fig. 1 illustrates how the exposure component of the health risk assessment implemented in the current study was modified for future climate scenarios. The following discussion will focus on the elements of the model that will be adjusted to allow for the incorporation of climate change projections. For detailed discussion of the original model development see the authors' previous work (Harmon O'Driscoll et al., 2024). This framework was applied to an Irish case study (Section 2.4), using standardised climate projections for Ireland as well as national datasets, to illustrate how this framework can be applied to assess the potential changes in pesticide concentrations in drinking water and resulting human health risks under different time horizons and climate scenarios. As this is an Irish case study, the influence of water treatment on pesticide concentrations were not considered, as effective pesticide removal processes are not currently implemented in water treatment plants within the Republic of Ireland (Éireann, 2021). However, the model was developed to be easily adapted to suit users' need and therefore reduction factors to account for pesticide removal processes may easily be incorporated into the model should this be appropriate for a given country.

Daily precipitation is the primary climatic parameter in the study's exposure model as it influences both the runoff volume (Eq. (1)) and the percentage of applied pesticide available for runoff. Given that climate change is likely to have an impact on precipitation intensity and frequency in Ireland (Nolan and Flanagan, 2020), there is significant potential for climate change to alter the level of risk associated with pesticide exposure.

First, daily runoff volume, Q (mm.day^{-1}), was calculated using model modifications adapted from the USDA's SCS Curve Number approach (USDA, 2004b):

$$Q = \frac{(R-0.2S)^2}{(R+0.8S)}; \text{for } R \geq 0.2\text{S} \tag{1}$$

$Q = 0$; for $R < 0.2$Swhere R is daily precipitation (mm.day^{-1}), and S is the maximum potential soil retention (mm). Based on Eq. (1), runoff will only occur when a precipitation event exceeds the threshold value of 20 % of the maximum potential soil retention. The threshold value varies based on the interaction between a site's land-use, soil type and cover conditions, as detailed in USDA (2004a). This highlights the importance of changes in precipitation patterns under future climates, as heavier precipitation periods are likely to result in more exceedances of this threshold and hence more frequent runoff events. In order to account for the changes to climate variables in the SFIL model, a number of climate-adjusted probability distributions were developed as follows (Fig. 1):

Climate change adjustment factors.

The gamma distribution was identified as the most appropriate for daily precipitation data based on distribution fitting to historical data and approaches in literature (Chandler and Wheater, 2002; Mockler et al., 2016; Ye et al., 2018). Daily precipitation under baseline conditions (R) was modelled as follows:

$$f(R) = \frac{\left(R^{\alpha-1} exp^{-\frac{R}{\beta}}\right)}{\beta^{\alpha}\Gamma(\alpha)}; \text{for } 0 < \text{R} < \infty \tag{2}$$

where α and β are the shape and scale parameters of the gamma distribution, and the gamma function Γ(α) is defined as:

$$\Gamma(\alpha) = \int_0^{\infty} R^{\alpha-1} exp^{-R}\, dR \tag{3}$$

In order to adapt Eq. (1) to account for the impacts of climate change on precipitation and resulting runoff, the shape and scale parameters (α and β) of Eq. (2) were modified to fit projected precipitation data (α_{cs} and β_{cs}, whereby cs is the different climate scenarios) as follows:

$$\alpha_{cs} = \frac{\mu^2}{\sigma^2} x \frac{\left(1 + \frac{\delta_{mean}(t)}{100}\right)^2}{\left(1 + \frac{\delta_{variance}(t)}{100}\right)^2} \tag{4}$$

$$\beta_{cs} = \frac{\sigma^2}{\mu} x \frac{\left(1 + \frac{\delta_{variance}(t)}{100}\right)^2}{1 + \delta_{mean}(t)} \tag{5}$$

where μ and σ are the mean and variance of baseline precipitation data, and $\delta_{mean}(t)$ and $\delta_{variance}(t)$ are timeframe-dependent percentage changes to the mean and variance value for each climate scenario (Table 1). Daily precipitation under future climate scenarios (R_{cs}) was then modelled based on Eq. (2) but using these adapted shape and scale parameters.

Eqs. (2)–(5) demonstrate how daily precipitation values for future climates were developed for the pesticide transport model. However, the resulting precipitation dataset is for the whole year, including months outside the growing season where there is little or no pesticide application. Previous work by the authors identified that predicted environmental concentration, and resulting health risk, decreased by up to 25 % when modelling for growing season period only, due to decreased levels of precipitation in the Irish growing season compared to annual conditions (Harmon O'Driscoll et al., 2024). As a result, pesticide risk can be overstated when using annual climatic conditions. Additionally, pesticide application timings are largely dependent on the timings of crop planting and growing during the growing season (Bareille et al., 2024). Therefore, this study only considered growing season conditions in the modelling of pesticide risk by limiting climate projections to this period only, thereby better representing realistic climatic conditions during pesticide application. The thermal growing season was used to assess the period of pesticide application, extrapolating growing season conditions from daily temperature projections as shown in Fig. 1. The thermal growing season (TGS, in days) is defined as the length of time between the first six successive days with an average temperature greater than five degrees, and the first six successive days with an average temperature less than five degrees (Dunn et al., 2020; Nolan and Flanagan, 2020), or as follows:

$$TGS = n - (i_2 - i_1)$$

$$where\ i_1 = min[i|(T_i > 5)\ and\ (T_{i+1} > 5) and \ldots (T_{i+5} > 5)]$$

$$\text{and } i_2 = min[i|i > i_1 + 5\ and\ (T_i < 5)\ and\ (T_{i+1} < 5)\ and \ldots (T_{i+5} < 5)] \tag{6}$$

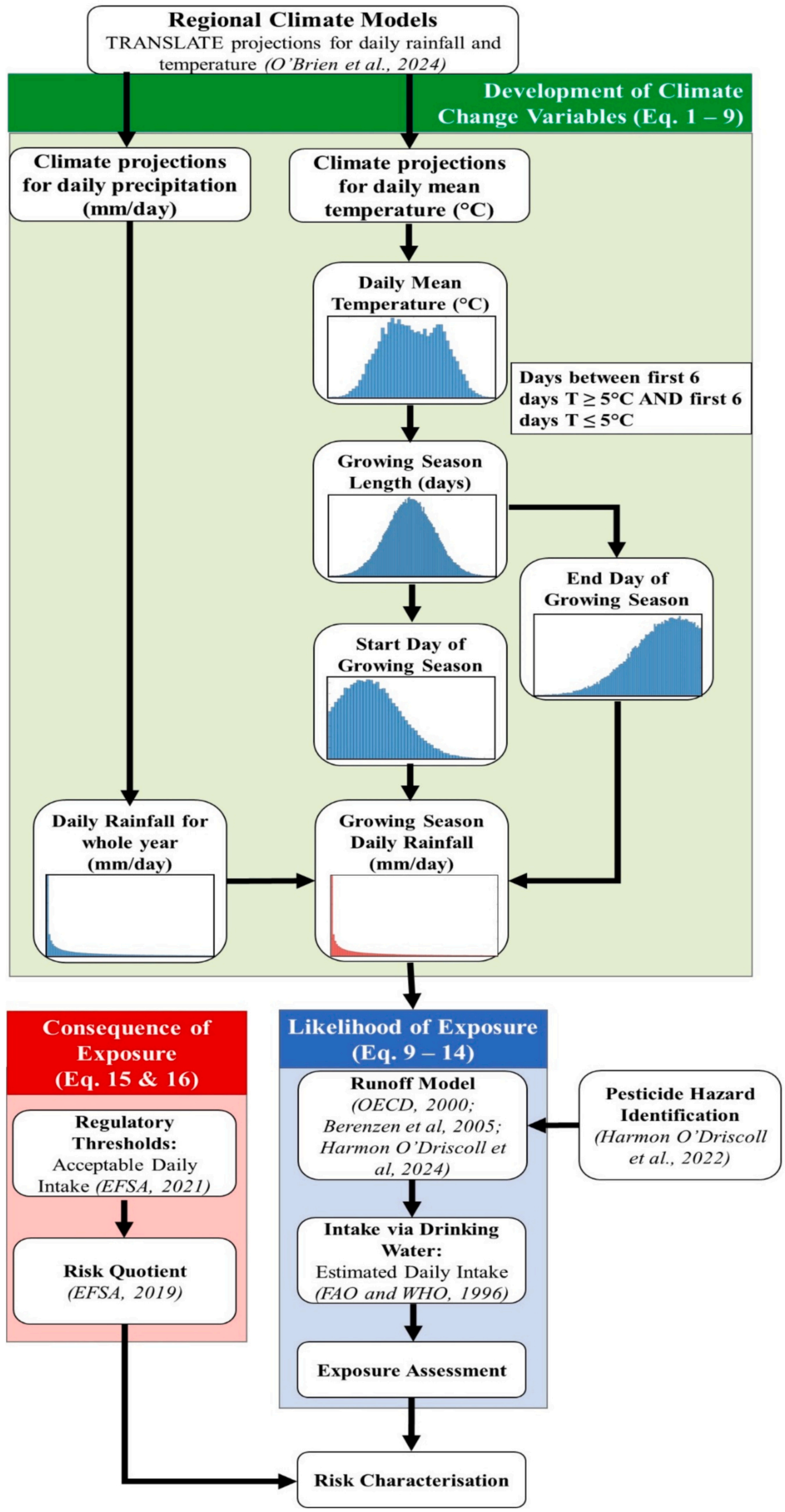

Fig. 1. Health risk assessment framework adapted for climate change.

where n is the number of days in the year, i_1 is the first day whereby the following five consecutive days have a mean temperature >5 °C, i_2 is the first day whereby the next five consecutive days have a mean temperature <5 °C, and T_i is the mean air temperatures (°C) on day i. Based on best-fit analysis of the observed daily mean temperature for baseline conditions, a normal distribution was used to obtain daily mean temperature:

$$f(T) = \frac{1}{\sqrt{2\pi\sigma^2}} exp^{-\frac{(T-\mu)^2}{2\sigma^2}} \tag{7}$$

where μ and σ are the mean and standard deviation of the observed daily temperature data for baseline conditions. A modification was made to Eq. (7) to allow for climate-related effects to be incorporated into modelling daily temperature values for future scenarios (T_{cs}) as follows:

$$f(T_{cs}) = \frac{1}{\sqrt{2\pi\left(\sigma\left(\frac{(1+\delta_{variance}(t))}{100}\right)\right)^2}} exp^{-\frac{\left(T_{cs}-\left(\mu\left(\frac{1+\delta_{mean}(t)}{100}\right)\right)\right)^2}{2\left(\sigma\left(\frac{(1+\delta_{variance}(t))}{100}\right)\right)^2}} \tag{8}$$

where $\delta_{mean}(t)$ and $\delta_{variance}(t)$ are timeframe-dependent percentage changes to the mean and variance value for each climate scenario (Table 1). For each climate scenario, the growing season length is calculated using Eq. (6); for future daily temperatures as obtained from Eq. (8), and distributions fitted to the resulting growing season length data. Additionally, distributions for the start and end day of growing seasons were extrapolated from growing season lengths, and these conditions were then used to extract daily precipitation projections for growing season conditions only from projected precipitation for the whole year for each of the five climate scenarios as shown in Fig. 1. The growing season conditions and resulting daily precipitation during the growing season for each simulated climate scenario are discussed in Section 3.1. The probability distributions for the climate change adjusted climate variables are then used in to calculate future pesticide concentrations and resulting human health risks (Eqs. (9)–(16)) for different time horizons and climates scenarios with findings for an Irish context presented in Section 3.2 and 3.3.

Climate-modified pesticide health risk model

Eq. (1) was then modified for runoff volume under future climates (Q_{cs}) to account for the adaptations applied to precipitation data as follows:

$$Q_{cs} = \frac{(R_{cs}-0.2S)^2}{(R_{cs}+0.8S)}; \text{ for } R_{cs} \geq 0.2S$$

$$Q_{cs} = 0; \text{ for } R_{cs} < 0.2S \tag{9}$$

Daily precipitation and daily runoff volume for the climate change scenario, as calculated in Eqs. (2) and (9), were then used to estimate the percentage of applied pesticide dose lost to runoff ($L_{\%}$), in combination with pesticide physiochemical properties and several site conditions, as follows:

$$L_{\%} = \frac{Q_{cs}}{R_{cs}} x \frac{exp^{\frac{-tln2}{DT_{50,s}}} x\, 100}{1+K_d} x f_1 x f_2 x f_3 \tag{10}$$

where $DT_{50,s}$ and K_d are input parameters based on pesticide properties (see Table S1), and f_1, f_2, and f_3 are correction factors to account for site conditions (Beinat and Van den Berg, 1996; OECD, 2000; FOCUS, 2002; Berenzen et al., 2005).

It has been found that variations in temperature can affect the rate of pesticide degradation in the environment (Gentil et al., 2020; Petrova et al., 2021; Campan et al., 2023). Therefore, an adjustment factor can be used to adapt pesticide half-lives for future climate conditions to account for how a warming climate may lead to a change in the rate of degradation. In this framework, users may apply a standard Arrhenius equation (FOCUS, 2011) to develop this adjusted half-life data as follows:

$$DT_{50,CS} = DT_{50}\, x \left(e^{\frac{E_a}{R} x \left(\frac{1}{T}-\frac{1}{T_{cs}}\right)}\right) \tag{11}$$

where $DT_{50,C}$ is the new pesticide half-life resulting from temperature projections for future scenarios, DT_{50} is the original pesticide half-life (days), E_a is the activation energy of the pesticide (kJ/mol K), R is the universal gas constant (8.31447 J/mol K). The reference temperature (T) is taken to be 20 °C or 293.2 K, as the half-life data was obtained from EFSA reports, where pesticide half-life is recorded at 20 °C (EFSA, 2021). T_{cs} is the projected temperature accounting for climate change which was obtained using Eqs. (7) and (8) and converted to Kelvin. This equation however may not be suitable in representing the increased rate in degradation in regions with cool climates, such as the mild temperate oceanic climate in Ireland, as daily temperatures even under future climates may not exceed the 20 °C reference temperature. In fact, the authors found that the increased daily temperatures lead to no, or very minor changes in predicted pesticide concentrations in Ireland due to the relatively low daily temperatures under all climate scenarios. Therefore, this was excluded from the analysis in Section 3. However, this factor may be important for warmer continental, tropical and Mediterranean climates with high daily temperatures, therefore users are advised to consider whether this factor should be considered for their own needs.

P_c, the pesticide concentration at the edge of field ($\mu g.l^{-1}$), was calculated as per Berenzen et al. (2005):

$$P_c = L_{\%} x P_a x \frac{1}{Q_{stream} x\, \Delta t} \tag{12}$$

whereby P_a is the application dose of the pesticide (μg), Q_{stream} is the flowrate of the waterbody ($l \cdot s^{-1}$) and Δt is the length of the precipitation event (s), as defined by APVMA (2020).

The predicted pesticide concentration in drinking water was assessed as the product of the edge-of-field concentration (P_c) and several in-stream reduction factors (ρ_{ss}, ρ_{sed}, ρ_{deg}, see Supplementary Information) (ECB, 2003; EPA NZ, 2020; Harmon O'Driscoll et al., 2024) that may reduce the concentration of pesticides over a retention period of five days (FOCUS, 2015):

$$PEC = P_c\, x\, \rho_{ss}\, x\, \rho_{sed}\, x\, \rho_{deg} \tag{13}$$

The resulting distribution of pesticide concentrations from Eq. (13) can be used to form the basis of a range of pesticide risk assessments. However, for the purpose of this study, modelled concentrations were used to calculate human exposure rate to a pesticide according to FAO and WHO (1997) guidelines:

$$EDI = \frac{PEC\, x\, WC}{BW\, x\, 1000} \tag{14}$$

where EDI is the estimated daily intake ($mg.kg^{-1}.day^{-1}$), WC is the rate of water consumption ($l.day^{-1}$), BW is bodyweight (kg), and 1000 is a conversion factor. The estimated daily intake is the final output of the exposure component of the risk modelling framework described in Fig. 1. To estimate the human health risk resulting from this exposure, it was then compared to the individual pesticide's acceptable daily intake (ADI) ($mg.kg^{-1}.day^{-1}$), the EFSA's regulatory thresholds for chronic health, in a risk quotient approach to provide a human health risk assessment (EFSA, 2019, 2021):

$$RQ = \frac{EDI}{ADI} \tag{15}$$

An additive approach, as recommended by the EFSA, was taken to assess the worst-case-scenario human health risk assuming exposure to a mixture of all modelled pesticides. Therefore, total health risk was calculated as the sum of risk quotients for each individual pesticide (i):

$$RQ_{total} = \sum RQ_i = \sum \frac{EDI_i}{ADI_i} \tag{16}$$

The proposed approach to adapting input variables in a probabilistic risk model for climate change projections may be applied using regional projections for any locations to suit users' needs. The use of the probabilistic approach described in Sections 2.3 enabled the model to consider the uncertainty and variability associated with the model, input parameters and importantly, climate projections for future scenarios.

2.3. Probabilistic modelling methodology

In developing the probabilistic approach used for the risk modelling framework in this study, input parameters, including climate variables under future scenarios, soil data, and pesticide properties, were fitted with probability distributions based on literature and best-fit analysis to available data. The probability distributions and statistical parameters developed for the input data are detailed in Section 2.4. A Monte-Carlo simulation method was then applied to populate the output distributions by repeatedly running the model with randomly selected inputs from the defined distributions. For each iteration of the model, one random sample was taken from each input distribution providing a distribution of model outputs. The model was run for 1,000,000 iterations for each location examined herein which was found to be more than enough to ensure statistical stability. This approach was taken in developing distributions for growing season conditions (Section 3.1), as well as estimating pesticide concentrations in drinking water and resulting health risk (Section 3.2 and 3.3). The use of Monte-Carlo simulations allowed incorporation of input data uncertainty and variability into the model, providing a distribution of estimated pesticide concentrations and resulting pesticide risk quotients, considering the effects of uncertainty and variability on model outputs.

2.4. Model parameterisation

A hypothetical agricultural site representing the average Irish conditions was developed by the authors in a previous study (Harmon O'Driscoll et al., 2024). It is similar to the site conditions at both locations in the current study, and so was used to represent their soil and agricultural conditions herein. Therefore, any differences in pesticide risks at the two locations will be a result of the regional variation of climate impacts as opposed to the influence of minor variations in site conditions. The influence of climate change on the climate statistical parameters used in assessing pesticide risk are discussed in Section 3.1.

The Irish Department of Agricultural, Food and the Marine has listed 82 pesticides used on Irish grassland and fodder based on the most recent pesticide usage surveys (DAFM, 2020, 2021, 2024). A previous risk screening study of these pesticides identified 15 key pesticides based on their mobility, toxicity to humans, quantity of use in Ireland, or a combination of three (Harmon O'Driscoll et al., 2022). Details of the pesticide properties, their classifications, and their statistical parameters used in the modelling process are available in Table S1. National population data for body weight and water consumption rates were obtained from the Irish Unversities Nutrional Alliance national surveys and USEPA guidelines to probabilistically modelling pesticide exposure rates (USEPA, 2004; IUNA, 2011, 2021). The statistical parameters developed for site and population data used in this study are presented in Table S2.

3. Results and discussion

3.1. Impact of climate change on growing season

Climate projections for Ireland indicate that daily temperatures will rise in the future, and this is likely to affect growing season conditions across the country which may have significant effects on the agricultural sector. At both study locations daily temperatures are projected to increase, with temperatures increases in the NE site ranging by 1.0 °C for RCP4.5 (2050) to 2.65 °C for RCP8.5 (2100) relative to baseline conditions, and 0.92 to 2.42 °C at the SW site for the same conditions, respectively. All modelled future climate scenario showed that rising temperature resulted in a longer growing season that begins earlier in the year (Fig. 2). Under baseline conditions, the average growing season length in the NE site is around 280 days, but this increases substantially for all future scenarios, with season length increasing to 310 days under RCP4.5 (2050) and to 345 day under RCP8.5 (2100) conditions, equivalent to a 7–19 % increase in growing season days. The same climate conditions result in an average 340–360 day long growing season in the SW site, compared to 320 days under baseline conditions (Fig. 2). These changes suggest that climate change will have a slightly greater impact on growing season in the north-east compared to the south-west, which is due to the larger increase in projected temperatures (Table 1). Despite climate change having a greater overall impact on the growing season in the NE site, the SW site has a longer agricultural season under all scenarios due to baseline conditions, as shown in Fig. 2.

Changes to growing season are important when considering pesticide risks as they define the period of the year when pesticides tend to be applied. In Ireland, under current climatic conditions accounting for sustained cold periods in inland regions, the average national growing season commonly starts in March and ends around November (Teagasc, 2017), therefore application often avoids the heavier precipitation associated with winter in the Northern Hemisphere (November – February in Ireland). However, as seen in the shift of the start date curves in Fig. 2, the growing season for all future climate scenarios is likely to start much earlier in the year. The start of the season shifts from the beginning of March in the north-east under baseline conditions, to mid-January under RCP8.5 (2100) conditions, and from the start of February to almost the beginning of the year in the SW site for the same scenarios. Therefore, the agricultural season is projected to start and end in winter months for the Ireland. Future application periods are therefore likely to coincide with these heavier daily precipitation events in Ireland as future daily winter precipitation is projected to have the greatest increase of the four seasons (Nolan and Flanagan, 2020). This is particularly likely for RCP8.5 (2100) conditions associated with the longest growing season projections in both locations, where the 50th percentile start and end date have shifted to early/mid January and mid/late December, respectively.

In the NE site, average precipitation across a growing season increases from 590 mm/season (baseline conditions) to 653 mm/season (RCP4.5 (2050)), and 788 mm/season (RCP8.5 (2100)), corresponding to a 3.6 % – 12 % increase in daily precipitation over a growing season. Similarly, in the SW site, daily average precipitation is projected to increase by 7.6 % – 16.5 % under the same climatic conditions relative to baseline conditions. This increase in precipitation intensity during periods of pesticide application will lead to greater runoff volumes, higher rates of pesticide exposure in drinking water supplies, and therefore increases in health risks. The findings indicate that there is regional variation in the changes to length of growing season and resulting daily precipitation over the growing season period, with the SW site likely to experience wetter growing season. Therefore, it is likely that there will be more opportunity for pesticide runoff in the SW site than the NE site which may lead to regional discrepancies in risk. This will be discussed in more detail in Section 3.3.

This analysis and the percentage changes in temperature, precipitation and growing season were used to inform the development of probability distributions for climate variables (Eqs. (2)–(8) in Section 2.2). Table 2 presents the developed climate parameters for each scenario, which were used in to predict changes in pesticide concentrations and resulting risks (Eqs. (9)–(16) in Section 2.2) under future climate scenarios at the two study as presented in Section 3.2 and Section 3.3.

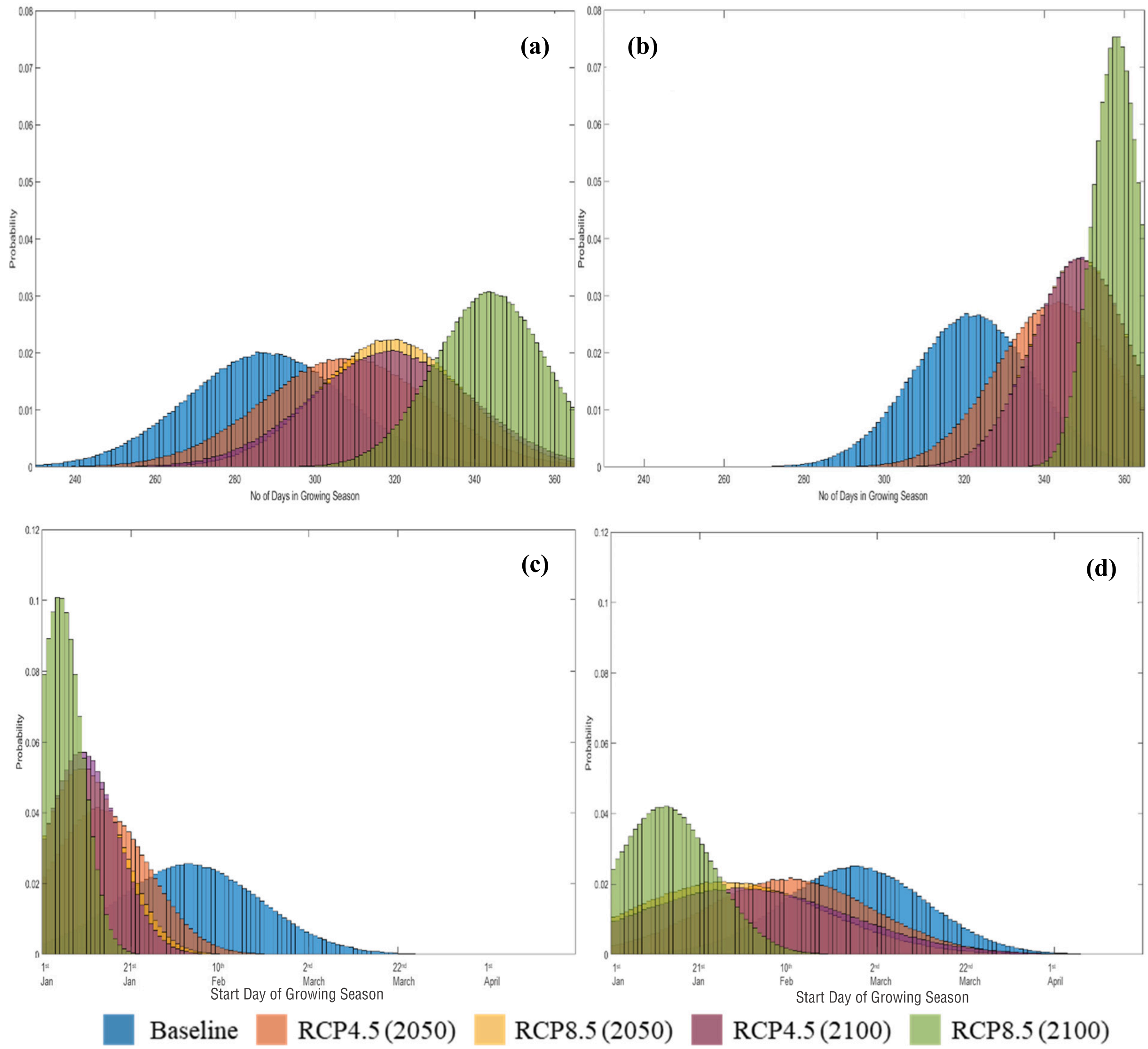

Fig. 2. (a) Distribution of growing season length for the five model scenarios in Dunleer (NE); (b) Distribution of growing season length for the five model scenarios in Causeway (SW); (c) Distribution of growing season start date for five model scenarios in Dunleer (NE) and (d) Distribution of growing season start date for five model scenarios in Causeway (SW).

Table 2
Statistical parameters for climatic parameters under the different climatic scenarios.

Scenario	Start of growing season[a]			Growing season length (days)			Growing season daily precipitation (mm.day^{-1})[b]		
	Distribution	μ	σ	Distribution	μ	σ	Distribution	a	b
Dunleer (North-East site)									
Baseline (1990)	Normal	55	16	Normal	288	20	Gamma	0.29	7.14
RCP4.5 (2050)	Normal	40	19	Normal	308	21	Gamma	0.27	7.76
RCP8.5 (2050)	Lognormal	26	22	Normal	319	18	Gamma	0.27	7.93
RCP4.5 (2100)	Lognormal	29	24	Normal	319	20	Gamma	0.27	8.14
RCP8.5 (2100)	Normal	12	11	Normal	344	14	Gamma	0.26	8.89
Causeway (South-West site)									
Baseline (1990)	Lognormal	33	16	Normal	322	15	Gamma	0.44	6.39
RCP4.5 (2050)	Lognormal	13	11	Normal	343	15	Gamma	0.37	8.05
RCP8.5 (2050)	Lognormal	9	9	Normal	349	12	Gamma	0.37	8.38
RCP4.5 (2100)	Lognormal	9	8	Normal	349	12	Gamma	0.37	8.43
RCP8.5 (2100)	Lognormal	4	5	Normal	358	6	Gamma	0.35	9.36

[a] Note: Day 1 = 1st January.
[b] Statistical parameters for daily precipitation consider the changes in growing season due to daily temperature projections, and the changes in daily precipitation due to projected precipitation for each climate scenario.

3.2. Impact of climate change on pesticide exposure and risk

As discussed in the introduction, there is a crucial gap in in current pesticide risk science as there is a lack of analysis on how climate change may influence pesticide contamination of drinking water and resulting human health effects. The following sections present the findings from an Irish case study to illustrate how the proposed methodology may be applied to quantify the climate change impacts on pesticide concentrations and pesticide-related human health risks, thereby provide risk-based information for decision making and adaptive practices. This analysis was conducted at two locations in Ireland. However, to facilitate detailed discussion of the results, this section will be focus on one location only (Causeway in the south-west) and will compare 2100 conditions to baseline conditions. Section 3.3 will present results for both locations to compare the influence climate change has on pesticide risk in both sites, thus illustrating the regional variability of climate change impacts. However, detailed results for all climate scenarios, and changes to exposure/risk in the north-east are provided in the Supplementary Information (Fig. S2 – S12). The results presented in this section are derived from the probabilistic modelling framework outlined in Section 2.2, with baseline conditions representing current pesticide exposure levels and future projections incorporating climate-adjusted factors (Eqs. (2)–(8)). These adjustments account for the combined effects of changes in temperature, growing season and precipitation as discussed in Section 3.1. By integrating these factors, the methodology quantifies how climate change influences pesticide concentrations in drinking water and the associated health risks.

Pesticide concentration in surface water due to runoff is strongly related to precipitation (Utami et al., 2020). However, precipitation must exceed 20 % of the soil's maximum potential retention for runoff to occur (Eq. (1)). Therefore, based on the soil conditions and land-use combination at study site, runoff will only occur when the precipitation threshold of 8.3 mm.day^{-1} is reached (Harmon O'Driscoll et al., 2024). Under baseline conditions, precipitation events that trigger runoff will occur on average 35 times a year. However, climate change is projected to increase daily precipitation intensity, with compounding effects due to the extension of the growing season. Therefore, the number of annual runoff events are projected to increase on average to 42 days under RCP4.5 and 44 days annually under RCP8.5 by 2100. While this will increase the rate of pesticide loss to drinking water supplies, it is important to note, that based on the model data, 88–90 % of simulated days are unlikely to receive sufficient precipitation to trigger a runoff event under any model scenario (i.e. reach the 8.3 mm. day^{-1} threshold) (USDA, 2004b; Harmon O'Driscoll et al., 2024). As a result, the results in Fig. 3 are the 95th percentile pesticide concentrations only, as median values equate to zero runoff, and zero pesticide concentrations.

Triclopyr has the highest simulated concentration for all scenarios modelled, followed by MCPA, 2,4-D and mecoprop (Fig. 3). These pesticides occur in the highest concentrations due to their very low adsorption coefficients (Syafrudin et al., 2021) (Table S1), making them unlikely to adsorb to soil, and therefore are more readily available for runoff. Additionally, these pesticides have relatively high application rates (EFSA, 2021), which increases the amount of pesticide available for transport. For both RCP4.5 and RCP8.5 projections, pesticide concentrations increase compared to current climates, with the highest concentrations occurring under RCP8.5 conditions. The EU's limit of 0.1 μg.l^{-1} for an individual pesticide in drinking water (European Commission, 2009) is exceeded by five pesticides based on current climatic conditions. Under both RCP4.5 and RCP8.5 conditions, half of the fifteen modelled pesticides are predicted to exceed this limit at the 95th percentile concentration (Fig. 3). The simulated total concentration of the modelled pesticides also exceeds the EU's legal limit of 0.5 μg.l^{-1} total concentration in all modelled scenarios. However, under RCP8.5 conditions, the total concentration is predicted to be six times the legal limit, compared to 1.5 times the limit under current climatic conditions (Fig. 3).

The effect of climate change on simulated pesticide concentrations becomes more evident when the percentage changes to predicted concentrations are compared. Under RCP8.5 (2100), the worst-case climate

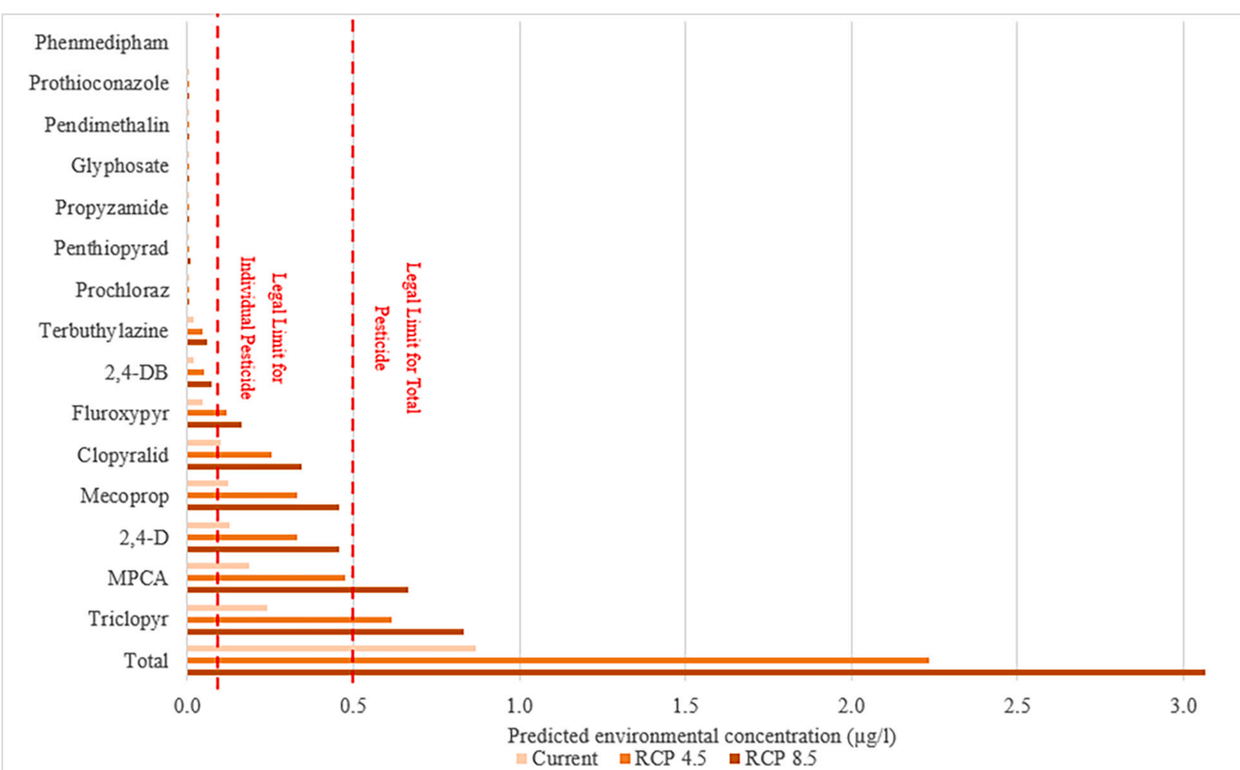

Fig. 3. 95th percentile predicted environmental concentration (μg.l^{-1}) for 2100 conditions in Causeway (SW).

scenario in this study, pesticide concentrations increased by an average of 36 %, compared to 24 % under RCP4.5 (2100) (Fig. S3). This can be explained by the differences in climate change impacts on model climate variables: under RCP8.5 (2100) conditions, mean daily precipitation during the growing season increases by over 16 %, and the resulting number of days where runoff threshold (8.3 $mm.day^{-1}$) is exceeded increases by 9 days compared to baseline conditions (an increase of 26 %). In comparison, mean precipitation increases by 11 % and the number of runoff events only increase by 14 % under RCP4.5 (2100). Therefore, under RCP 8.5 (2100), not only are there more runoff events than other modelled scenarios, but the resulting level of runoff is greater due to the projected increase in precipitation. As a result, the rate of exposure also increases substantially for both adults and children as can be seen in Tables S3 and S4. This significant increase in exposure leads to higher levels of health risks in both adults and children under future climatic conditions as shown in Fig. 4.

Despite the increase in EU drinking water quality limits exceedances and pesticide exposure rates, the resulting level of exposure was found to remain well below the acceptable daily intake under future climate scenarios. Therefore, health risks were found to be very low for all modelled pesticides (Fig. 4). The 99th percentile risk quotients for the fifteen modelled pesticides are well below a risk quotient of one. Therefore, based on this widely utilised metric, the risk of adverse health effects is extremely low for all modelled pesticides (EFSA, 2019). Mecoprop posed the highest risk to both adults and children as it has potential to cause developmental issues and therefore has a low acceptable daily intake value (0.01 $mg.kg^{-1}.day^{-1}$) (EFSA, 2021). The second highest risk level was attributed to triclopyr, which has a higher allowable daily intake (0.03 $mg.kg^{-1}.day^{-1}$) than mecoprop but has much higher simulated concentrations (Fig. 4). Despite the increasing likelihood of exposure to pesticides due to climate change effects, the resulting level of health risks posed to humans is still very low even under the most extreme climate scenario (RCP8.5 2100). Furthermore, the total risk from exposure to pesticides, assuming an additive effect of all modelled pesticides (EFSA, 2019), is also well below acceptable levels, ranging from a minimum of 0.026 for adults under baseline conditions to 0.080 for children under RCP8.5 (2100).

In fact, it was found that there was a less than a 0.01 % chance of exceeding an unacceptable level of risk, with 99.7 % of model simulations for mecoprop, the pesticide with the highest risk quotient in Fig. 5, producing a risk quotient <0.1. Although levels of risks are relatively low for all climate scenarios, climate change will influence the level of risk arising from pesticide exposure as pesticide risk was found to increase by an average of 24 % (RCP4.5 2100) to 37 % (RCP8.5 2100) across the various pesticides, relative to current conditions (Fig. 5). Due to the strong relationship between exposure rate and resulting health risk, RCP8.5 (2100) was shown to potentially have the greatest influence on pesticide health risk. The impact that climate change has on both the rate of pesticide loss to runoff and resulting health risk highlights the need for awareness in terms of agricultural practices and behaviours. For example, it has been shown that climate change will likely extend the growing season period for crops. This may also increase the abundance of pests, such as insects and fungi, and therefore cause an increase in pesticide usage or change in active ingredient for more effective treatment (Gagnon et al., 2016). Therefore, it is important to factor in climate change impacts in risk assessments, especially when land use or pesticide use may change into the future.

3.3. Regional variability in climate effects

The previous section focused on the changes in pesticide concentrations at one study location, however climate variable data in Section 2.1 and findings in Section 3.1 highlight the significant regional variation in climate effects at the two locations. Existing pesticide-related environmental risk studies tend to use global climate change models or lower-resolution climate projections in their analysis of climate change effects which cannot account for this fine spatial variability, for example Martínez-Megías et al. (2023) used projections from the Max Plank Institute Earth System Model at a base resolution of 200 km (MPI-ESM LR, DKRZ (2013)). However, as discussed in the introduction, low resolution projections cannot account for the highly spatially variable effects of climate change (Kotamarthi et al., 2021) and the lack of

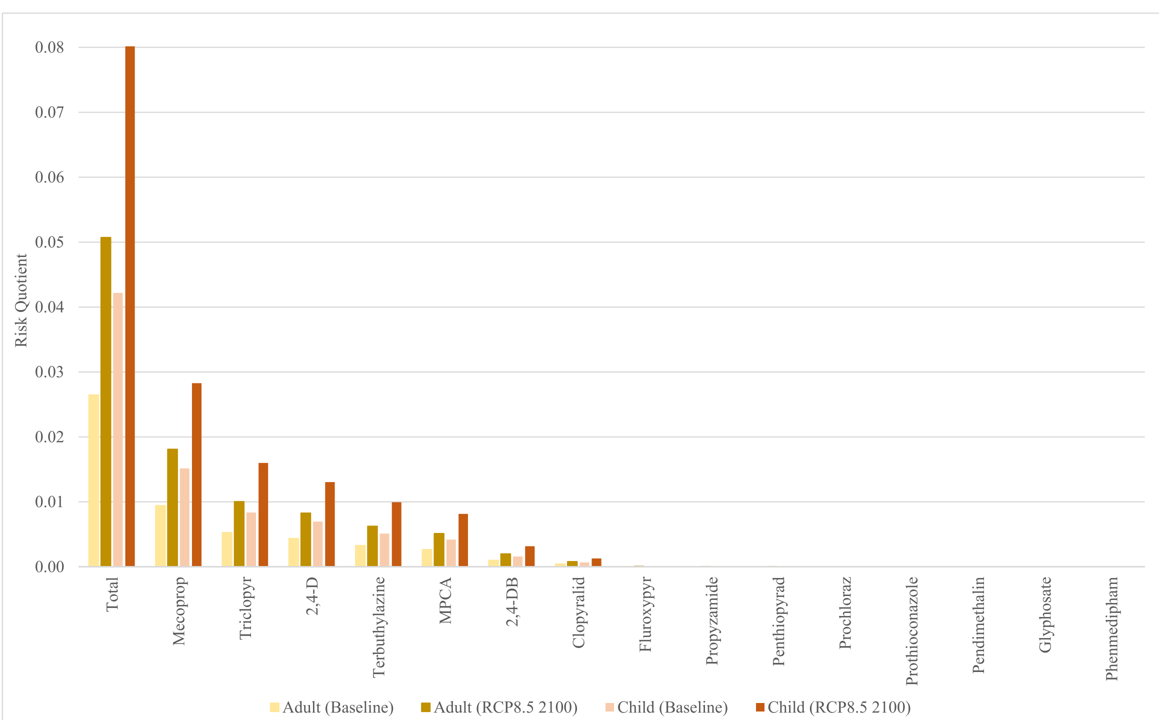

Fig. 4. 99th percentile pesticide risk level for adults and children for baseline and RCP8.5 (2100) conditions in Causeway (SW).

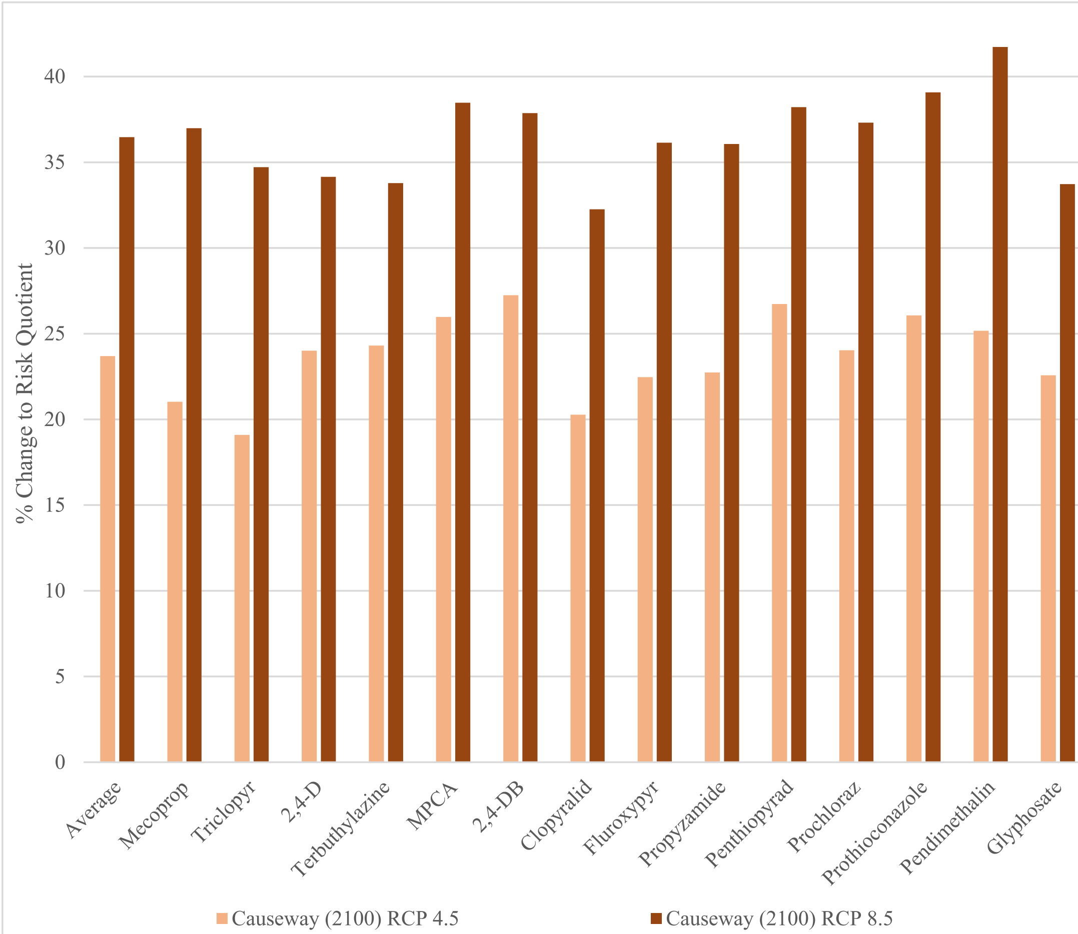

Fig. 5. Percentage change 99th percentile risk quotient (child) for future scenarios relative to current climate.

regionally specific risk assessments can limit the cost-effectiveness and resource efficiency of climate action and adaptation (Ryan and Stewart, 2020). Therefore, this section analyses climate change impacts at a local scale using high-resolution climate projections (at a 4 km resolution) to investigate whether climate change will result in regional-specific changes to pesticide-related risks. To facilitate concise but detailed

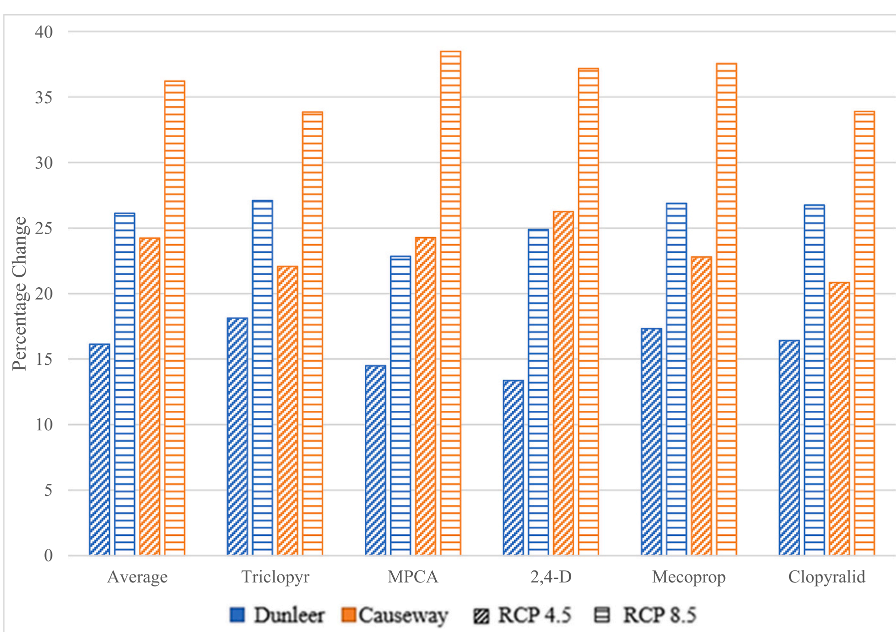

Fig. 6. Percentage change to 99th percentile RQ (child) for 2100 relative to current in Dunleer (NE) and Causeway (SW).

discussion of the results, Fig. 6 presents the percentage increase in pesticide risk for only five pesticides, which were identified as the highest risk pesticides (Fig. 4), under RCP4.5 and RCP8.5 (2100) (See Fig. S12 for all modelled pesticides).

In the NE site, the influence of climate change results in an average 17 % increase in health risks under RCP4.5, and a 26 % increase under RCP8.5 conditions by the end of the century across the five pesticides. However, health risks from pesticides in drinking water are on average 50 % higher in the SW site than the NE site under baseline conditions and increase to almost 90 % higher under RCP8.5 (2100) conditions (Fig. S11). The differences in climate impacts across the two locations can be partly explained by considering the influence of baseline conditions on the runoff model. Firstly, baseline daily temperatures are higher in the SW ($\mu = 10.6$ °C) compared to the NE ($\mu = 9.9$ °C), and as a result, the growing season is on average 34 days longer in the SW than the NE. Therefore, the likelihood of pesticide application during periods of wetter weather for baseline conditions is higher in the SW, as previously discussed. This impact is amplified under RCP8.5, whereby the growing season length is effectively the entire year in the SW site. More importantly, however, under baseline conditions, the south-west of Ireland experiences significantly more precipitation than the north-east (1074 mm/y versus 785 mm/y), as well as heavier precipitation days. This regional variability is predicted to intensify in the future, with climate projections showing a greater increase in heavy precipitation periods along the west coast of Ireland (Causeway area) than in eastern regions (Dunleer area) (O'Loughlin and Mozafari, 2023; O'Brien et al., 2024). This is particularly important due to the bimodal relationship between precipitation and runoff, where runoff only occurs during periods of heavy precipitation i.e. 8.3 mm.day^{-1} for the site conditions considered above. The projected changes to precipitation in the SW site result in an increase in the number of days where runoff may occur, rising from 35 days under baseline conditions to 42 and 44 days for RCP4.5 and RCP8.5, respectively. The NE site, however, experiences far fewer runoff events, with precipitation exceedances occurring only 24 times a year on average for baseline conditions, increasing to a maximum of 30 under RCP8.5 (2100). Additionally, the volume of runoff is also much less, as overall daily precipitation in the south-west is much greater than the north-east under baseline and future climatic conditions. Average daily precipitation during the growing season in the NE site increases from a baseline of 2.05 mm.day^{-1} by a maximum of 11.7 % (2.29 mm.day^{-1}, RCP8.5 (2100)). In the SW site, daily precipitation increases from an average of 2.78 mm.day^{-1} by 16.5 % (3.24 mm.day^{-1}, RCP8.5 (2100)). Therefore, although climate projections suggest that climate change has a slightly greater impact on climate variables in Dunleer (Table 2), the changes to climate variables resulting from climate change have a greater impact on pesticide runoff and resulting health risks in the SW site. The results indicate that a national adaptation strategy based on a single-location could under/overestimate pesticide-related health risks at a local scale and therefore high-resolution climate projections to account for the spatial variability of climate effects and illustrates that. Consequently, this analysis emphasises the need to carry out localised assessment of climate impacts on pesticide risks.

4. Conclusions

This study provides users with a practical and accessible predictive model framework for the integration of downscaled, bias-corrected climate projections into a quantitative human health risk assessment, providing an anticipatory analysis of pesticide-related human health risks, an area which is currently overlooked in pesticide risk science. Climate projections for the study locations indicated an increase in average daily precipitation and temperature, in line with increases across the North Atlantic region, which will likely affect pesticide concentrations in surface waters. A case study of 15 pesticides in two locations in Ireland was used to illustrate the application of the proposed framework and to provide insights into how pesticide exposure and associated health risks may change under future climates. Resulting human health risks were projected to increase for all climate scenarios, with the greatest increase in health risks at both study sites occurring under RCP8.5 (2100). The findings demonstrate that increased precipitation intensity, coupled with extended growing seasons due to higher daily temperatures, will likely elevate pesticide runoff and contamination risks, particularly in regions with already high precipitation levels. However, despite the projected rise in exposure, estimated human health risk remained below EU regulatory health thresholds, indicating that while climate change will intensify pesticide contamination in drinking water, the likelihood of adverse human health effects remains low under the modelled scenarios. The integration of high-resolution climate projections allowed for detailed regional comparison of climate change impacts on pesticide exposure and risks, addressing a recognised need for localised risk assessment for climate adaptation. Results indicated significant regional variability, with climate change shown to result in greater pesticide risks in the south-west due to compared to the north-east. This underscores the potential shortcomings of national-scale climate adaptation strategies that do not account for regional differences, emphasizing the need for localised assessments to optimise risk mitigation efforts.

Despite the low predicted risks, this study highlights the need to account for climate change impacts in pesticide risk assessments to inform adaptation and mitigation strategies for the future where necessary. As a result, this study has wide reaching implications for stakeholders, policymakers and pesticide research. For example, the findings illustrate that climate change may increase pesticide human health risks beyond which is not currently considered in EU regulations. Therefore, agricultural policymakers should consider incorporating climate change assessments to future-proof their regulation. Additionally, climate adaptation strategies should consider developing local/regional strategies, as discussed previously, that may provide more targeted responses to climate change impacts and enable more efficient use of resources. Furthermore, while findings in the presented case study suggest that overall increases in human health risks remain too low to require mitigation, the results also highlight that climate change has the potential to impact pesticides risks and potentially risk associated with other contaminants. Therefore, water utilities should consider incorporating climate projections into their assessments to enable future-planning of water quality management and infrastructure investment plans.

This study contributes to the literature quantifying the impacts of climate change on health risks by demonstrating how multi-ensemble, bias-corrected projections from downscaled regional climate models may be incorporated into probabilistic models, using a practical approach suitable for international users. However, this study is somewhat limited in its representation of future climate scenarios due to the focus only on quantifying the direct impacts of climate change on pesticide risk by considering changes to precipitation and temperature. There is also a need to expand the research to consider the indirect effects of climate change, including changes to land productivity, pest prevalence, and changes to agricultural policy and land-use, which may have a compounding effect on pesticides risks. These factors will likely affect land-use, thereby changing types of pesticides in use, as well as increasing/decreasing pesticide application rates in response to pest prevalence and policy changes. Therefore, it is important that influence on risks arising from agrochemicals be considered in climate adaptation strategies especially in terms of future public health and environmental health strategies. Furthermore, this study provided a first step towards probabilistic assessment of human health risks under future climates, however it relied on the commonly used risk quotient at the risk characterisation stage to quantify health risk levels which does not enable detailed analysis of the uncertainty associated with pesticide toxicity. Alternative health impact models for risk characterisation could also be incorporated into this approach to improve representation of health impacts in detailed analysis and provide more precise information regarding health impacts arising from pesticide exposure.

CRediT authorship contribution statement

J. Harmon O'Driscoll: Writing – review & editing, Writing – original draft, Software, Methodology, Investigation, Formal analysis. **M.G. Healy:** Writing – review & editing, Supervision, Project administration, Funding acquisition, Conceptualization. **A. Siggins:** Writing – review & editing, Project administration, Funding acquisition. **J. McGinley:** Writing – review & editing. **E. O'Brien:** Writing – review & editing, Software, Data curation. **J. Wang:** Writing – review & editing, Software. **P. Holloway:** Writing – review & editing, Funding acquisition. **P.-E. Mellander:** Writing – review & editing. **L. Morrison:** Writing – review & editing. **S. Scannell:** Writing – review & editing. **P.C. Ryan:** Writing – review & editing, Writing – original draft, Supervision, Project administration, Funding acquisition, Conceptualization.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.scitotenv.2025.179090.

Data availability

Data will be made available on request.

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