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Assessing the impact of agricultural pesticides in the environment (phase II)

Science Report: SC030189/SR1

SAFETY FOR PEOPLE AND THE ENVIRONMENT

SCHO1206BMCU-E-P

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Our work includes tackling flooding and pollution incidents, reducing industry's impacts on the environment, cleaning up rivers, coastal waters and contaminated land, and improving wildlife habitats.

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Author(s):
Colin Brown, Sabine Beulke, Jeremy Biggs, Chris Holmes, Lorraine Maltby, Wendy van Beinum, Ryan Williams, Marian Yallop

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Research Contractor:
Central Science Laboratory, Sand Hutton, York, YO41 1LZ, UK
T: +44 (0)1904 462000

Environment Agency's Project Manager:
Emma Pemberton and Claire Wells
Environment Agency Science Group, Evenlode House, Howbery Park, Wallingford, OXON, OX10 8BD

Collaborator(s):
Pesticides Safety Directorate, Mallard House, Kings Pool, York, YO1 7PX

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Steve Killeen
Head of Science

Executive Summary

Pesticides are widely used in agriculture to protect crops from pests and diseases and to increase yield. As a result of their use, pesticides are routinely found in streams and rivers throughout the UK.

We know that the high concentrations of pesticides in water caused by pollution incidents can cause harmful effects and even kill aquatic plants and animals. We do not know if the lower concentrations of pesticides which enter streams and rivers as a result of normal agricultural use cause harm because of a lack of field based evidence. We need to know whether the current risk assessment and risk management procedures that we have in place for pesticides are appropriate and targeted at real environmental problems.

The overall aim of this project was to establish whether the normal use of agricultural pesticides leads to measurable adverse effects on aquatic invertebrates and plants. From this, the Environment Agency could develop a risk-based monitoring strategy for surface waters and provide field-based evidence to the Pesticides Safety Directorate (PSD) on whether the way that products are currently approved and used is precautionary.

The project was broken into smaller tasks, the first of which, reported here, underpins the later monitoring phase of this work, by identifying sites vulnerable to pesticide contamination and designing a sampling programme.

National-level risk mapping showed that crop type is the major driver for potential risk to the aquatic environment from agricultural use of pesticides. Orchards were identified as the crop type associated with the greatest potential risk. Crops more widely grown in England and Wales, such as cereals, oilseeds, potatoes and vegetables were found to be an order of magnitude smaller in risk than orchards.

Landscape analysis using Geographical Information System (GIS) datasets and aerial imagery investigated the local characteristics of orchard cultivation for almost 1,500 individual stream segments adjacent to orchards in Herefordshire, Kent and East Anglia. Coupled with site visits, the analysis showed that conditions associated with the greatest pesticide exposure occurred very infrequently.

Statistical analysis suggested that only relatively large effects of pesticides on aquatic communities would be detected using the empirical monitoring-based design. In locations where the greatest exposure to pesticides was anticipated, aquatic ecosystems generally lacked pesticide-sensitive organisms. Large, detectable changes in aquatic populations were therefore unlikely to occur. In addition, independent tests of passive sampling devices, the proposed chemical monitoring methodology for the project, showed that there were some gaps in the suite of chemicals that could be monitored and that some exposures would pass undetected because of their transient nature.

The decision was therefore taken to terminate the project at the end of this design phase. Despite this, the question to be addressed – namely, establishing the significance of effects of pesticide use on aquatic life - remains extremely important from both a policy and scientific perspective. This report makes recommendations for an experimental study that might be considered for future research activities by the Environment Agency and PSD.

Contents

1. Introduction

1.1 Background

Environmental regulators need to know whether the current risk assessment and risk management procedures that are in place for pesticides in the UK are appropriate and targeted at real environmental problems. Despite the vast array of studies reporting the presence of pesticides in surface waters there is currently insufficient field based scientific evidence to establish whether the use of pesticides for plant protection in agriculture adversely affects aquatic flora and fauna. In response to this issue, a review of scientific literature on the effects of pesticides on the aquatic environment was commissioned by the Environment Agency (Crane *et al*., 2003).

This review explored the published literature for evidence of effects from pesticides both in the UK and elsewhere. The main conclusions were that:

- relevant information is sparse, both for the UK and for other countries;
- only a few studies show that the potential for adverse effects exists, but it is not possible to determine from these whether this is a widespread or severe problem for the UK;
- biological and chemical data collected by the Environment Agency is not adequate to determine whether pesticides are causing adverse effects in UK surface waters.

The 2003 report recommended that the Environment Agency commission an extensive study to determine whether, and to what extent, agricultural pesticides impact on aquatic communities. A study design was suggested based on approaches used successfully in previous studies to establish links between pesticide exposure and detrimental effects. This report describes the first phase of research undertaken following this recommendation.

Subsequent to the report of Crane *et al*. (2003), an EPiF (Effects of Pesticides in the Field) workshop was convened in November 2003 to review available data in the EU on the ecological effects of pesticides. The aim of the workshop was to consider to what extent the available data quantified the exposure and effects of plant protection products on the environment and to examine whether these effects (or lack thereof) would be predicted by current risk assessment procedures. Separate subgroups comprising experts from regulatory authorities, the research community and industry considered evidence for field effects on aquatic organisms, terrestrial plants and invertebrates and terrestrial birds and mammals. The objectives of the workshop differed somewhat from those of Crane *et al*. (2003) in seeking a Europe-wide evaluation and comparing risk assessment with observed effects. Nevertheless, the workshop and review findings were broadly in agreement.

The final report from the EPiF workshop includes summaries of the discussions for each environmental compartment (Liess *et al*., 2005). The aquatic group concluded that it is widely acknowledged that farming (including land management, pesticide and fertiliser use, irrigation etc.) has an impact on water bodies in agronomic landscapes. Furthermore, monitoring studies presented in the workshop showed that there can be effects on non-target aquatic organisms in the field arising from pesticide use, although the studies concentrated on insecticides and macroinvertebrates (Liess and Schulz, 1999; Schulz *et al*., 2002). In addition, there is evidence of macrophyte loss from the agricultural landscape (Williams *et al*., 1998), and preliminary reports of the feminisation of male frogs in areas of high herbicide use (Hayes *et al*., 2002). The group also concluded that several fundamental issues remain:

- there are relatively few examples of field studies and little information on how widespread the observed effects are;
- there is uncertainty over the magnitude of exposure which would produce these effects;
- it is difficult to attribute a particular effect solely to pesticides, as the effects could also be the result of other stresses to the ecosystem.

1.2 Aims of the study

The aim of this project was to establish whether the use of agricultural pesticides is associated with measurable adverse effects on aquatic invertebrates and plants. From this, the Environment Agency could develop a risk-based monitoring strategy for pesticides in surface waters and provide field-based evidence to the Pesticides Safety Directorate (PSD) on whether the way that products are currently approved and used is precautionary.

The programme of work was divided into four tasks. Task 1, which is reported here, was to prepare for the monitoring phase of the work by identifying sites vulnerable to pesticide contamination and designing a sampling programme. The specific objectives of Task 1 were to:

- identify water bodies potentially vulnerable to pesticide contamination in at least one Environment Agency region;
- select a representative sample of these water bodies and design a strategy for sampling from these sites.

Progression to Tasks 2-4 depended on the results from Task 1. These tasks are not discussed further here, but would comprise the following:

Task 2 - sampling and analysis:

- biological monitoring (benthic invertebrates, benthic diatoms, macrophytes);
- chemical sampling;
- in situ approaches for bioassays and bioaccumulation;
- supporting measurements of physical factors.

Task 3 - data analysis, interpretation and laboratory confirmation of plausibility:

- data analysis and interpretation;
- laboratory confirmation of plausibility;
- conclusions, recommendations and preparation of report.

Task 4 would involve extrapolating the results.

2 Methods

2.1 Overview of risk mapping

The first objective of the project was to identify water bodies potentially vulnerable to pesticide contamination. A subset of the most vulnerable water bodies would then be selected for monitoring during the field phase of the project. It was decided that monitoring should be undertaken in small streams close to the source, in order to minimise the influence of confounding factors associated with other human activities. Thus, the assessment focused on streams but also considered ditches in low-lying areas where the stream network was not extensive.

A tiered approach with two levels of analysis was adopted for the risk mapping and identification of streams. First, national-level analysis was used to calculate the potential risk to aquatic organisms from exposure to pesticides and was mapped for the whole of England. Separate national-level maps were generated to show potential risks to aquatic invertebrates (*Daphnia magna)* and algae from agricultural pesticides. Calculations of exposure via spray drift and drain flow were combined with acute toxicity values to generate toxic units. These calculations were performed for all combinations of crops (15 categories), hydrogeological landscapes (10 categories) and regions (6 categories) and were based on pesticide use statistics averaged by crop/region combination.

Risk was then mapped at the catchment level by considering the relative contribution of the different crop-landscape-region combinations to the catchment. This approach could not be used for Wales because of a lack of harmonised cropping information. Arable and orchard crop cultivation is generally less intensive in Wales than in parts of England, and the exclusion of Wales from the mapping process is unlikely to have influenced the main results of the work.

From the results of this national analysis, three discrete areas were chosen to explore the potential exposure of streams at the landscape level. Areas of five by five km were selected based on the intensity of priority crop types adjacent to water. All individual water bodies within each block were examined using high-resolution spatial data to determine the length of stream perimeter potentially exposed, the degree to which it may be exposed to spray drift, and the presence, width and composition of any natural buffers. Thus, the nature and abundance of locations at greatest risk of impacts arising from the use of pesticides was established.

2.1.1 Datasets used

This section briefly describes the datasets used in this study.

Pesticide usage data

Pesticide usage data were taken from Department for Environment, Food and Rural Affairs (Defra) surveys (data collected between 2000-2003 according to crop type). Data are aggregated to six regions for England and Wales and were aggregated to 15 primary crop groupings for the purposes of this project.

Ecotoxicological data

An ecotoxicological database was created with acute toxicity LC50 and EC50 data for *Daphnia magna* and aquatic algae (usually *Scenedesmus*). Data were collated from the p-EMA database

(Lewis *et al*., 2003), from Agritox (www.inra.fr/agritox) and from the US Environment Protection Agency (US EPA) database (www.epa.gov/ecotox).

Aquatic landscapes

Aquatic landscape data were developed under Defra Project PN0931. Hydrogeological landscape classes define ten categories of landscape related to aquatic ecosystems (Table 2.1). Coverage exists for all of England, Scotland and Wales (Brown *et al*., 2006).

Table 2.1: Agricultural landscapes in England and Wales identified for Defra project PN0931

No.	Landscape	Total area (km^2)
1	River floodplains and low terraces	7,781
2	Warplands, fenlands and associated low terraces	9,017
3	Sandlands	10,871
4	Till landscapes (eutrophic)	22,151
5	Till landscapes (oligotrophic)	15,449
6	Pre-Quaternary clay landscapes	19,706
7	Chalk and limestone plateaux and coombe valleys	14,197
8	Pre-Quaternary loam landscapes	10,072
9	Mixed, hard, fissured rock and clay landscapes	12,259
10	Hard rock landscapes	23,342

Land Cover Map 2000

Twenty-five metre resolution land cover data developed by the Centre for Ecology and Hydrology are available across the UK. The LCM 2000 dataset classifies broad land cover types and enables the user to determine where within a ward agricultural production is present, and how to allocate ward-level crop statistics to each catchment (Land Cover Map 2000, CEH, © Crown copyright). Land cover classes used from the LCM 2000 data were arable cereals, arable horticulture, and arable non-rotational, since all crops of interest could be assigned to one of these three classifications.

Ward-level agricultural statistics from 2003 linked to ward geometry

Crop production statistics were used to estimate individual crop production within each catchment. These statistics are the result of the June 2003 ward-level agriculture statistics survey (www.defra.gov.uk/esg/work_htm/publications/cs/farmstats_web/Publications/complete_pubs.htm).

Catchments

Water Framework Directive river catchments - approximately 6,600 catchments in England and Wales, derived by the Environment Agency from 50-metre elevation data and corrected to 1:50,000 scale hydrology from the Centre for Ecology and Hydrology.

OS MasterMap® land cover classification

Detailed hydrology, land cover and imagery Ordnance Survey Mastermap data covering selected areas (hydrology and orchard boundaries) was used to select areas identified in the national analysis. Water features are represented as either lines for small streams and canals, or polygons for ponds, lakes, or rivers. Land cover types are classified as polygons.

For the purposes of this study, all lines and polygons attributed as 'inland water' were extracted from the full OS MasterMap® land cover. Line features attributed as 'inland water' were used to quantify hydrologic density in specific areas and indicate the presence of water when water was digitised on the aerial imagery.

All polygons attributed as ‘orchards’ or any mix of ‘orchards’ and other land cover were extracted. These polygons were used to select water features within 20 m for further analysis, and were also used to quantify land cover classified as orchards for landscape-level analyses.

OS MasterMap® aerial imagery

Aerial imagery (25 cm, colour) was used to generate highly accurate information on the proximity of orchards to water and the composition of the buffers (© Crown copyright). Imagery was provided for specific areas considered in the landscape-level analysis. Capture dates ranged from May 1999 to July 2003. Source images had been georeferenced in advance by the data provider.

2.2 National-level mapping

2.2.1 Calculation of risk from pesticide use by crop, hydrogeological landscape and region

Toxicity data

Biological monitoring during the field phase of the project was to focus primarily on effects on aquatic invertebrates and diatoms. Accordingly, indicator species for these two categories of organism were selected and used as the basis for risk calculation. *Daphnia magna* is the standard test species representing invertebrates in first tier risk assessment for pesticides, so acute toxicity data for this species were collated as an indicator of toxicity to invertebrates. Data for toxicity to algae (generally *Scenedesmus*) were collated as a surrogate for diatoms. A database compiled by the Central Science Laboratory (CSL) in consultation with PSD for a research project on aquatic indicators (Pesticides Forum paper PF83, 13 October 1999) was used as a base, with gaps filled using the p-EMA database (Lewis *et al*., 2003), Agritox (www.inra.fr/agritox) and databases from the US EPA (www.epa.gov/ecotox). Data were not available for a small number of compounds and here the toxicity value was taken from the pesticide from the same group with the closest structural similarity.

The aquatic indicators project included information on the regulatory status of the compound (for example, two compounds could have the same risk but would be ranked differently if one had just undergone re-registration). This was not considered necessary for this study and only the intrinsic hazard represented by the standard test on *Daphnia* or algae was taken into the risk assessment. This provided a baseline assessment whereby all chemicals are treated as equal, regardless of their period of development or the extent of higher tier work undertaken to support registration.

Crops/pesticide usage

Ward-level data from the Defra agricultural census were used to supplement georeferenced land use data from the Land Cover Map. Census data are available as spatial data layers for 26 individual crop types. Some categories are very small (such as turnips), so crops were aggregated into 15 major categories (Table 2.2). This aggregation was done by the Pesticide Usage Team at CSL to ensure that pesticide usage was relatively consistent within the categories.

Cropping information has been held separately for Wales since devolution. A census was undertaken in Wales in 2000, but the data are not in a suitable format to include in the mapping exercise. Risk maps were thus only generated for England.

Table 2.2: Crops for which toxic units were calculated

Crop category for risk calculation	Agriculture census categories
Beans (field)	Field beans
Bulbs	Bulbs and flowers
Cereals	Wheat + winter barley + spring barley + oats + other cereals
Hardy nursery stock	Hardy nursery
Hops	Separate processing of horticulture
Maize	Maize
Oilseeds	Oilseed rape and linseed
Peas and beans (veg)	Peas and beans
Peas (dry)	Peas (harvest dry)
Potatoes	Potatoes
Other stockfeed	Other crops for stockfeed
Other vegetables	Total vegetables in the open minus peas and beans
Soft fruit	Small fruit
Sugar beet	Sugar beet
Top fruit	Top fruit

Usage statistics were aggregated for the crop categories identified in Table 2.2 (Dean *et al*., 2001; Garthwaite *et al*., 2001, 2003, 2004; Garthwaite and Thomas, 2003a, 2003b, 2004). All statistics were expressed at regional level (six Defra regions - Northern, Eastern, South-Eastern, South-Western, Midlands and Western, Wales) to ensure that regional differences in pesticide use were incorporated into the mapping. All data were expressed as the average actual rate per unit area of crop. Timing of application was extracted from the usage database and used to determine whether or not transport via drain flow would be expected. Repeat applications of the same active ingredient to the same crop were combined into a single larger application. The date of the most recent survey varied between 2000 and 2003 (the latest for arable crops was 2002 at the time the mapping was undertaken). The area of some crops was extremely limited (for example, the total area of soft fruit in England and Wales in 2001 was 7,600 hectares), but all crops were included to provide total national coverage. The end result was a list of average usage per unit area for individual pesticides by crop category and region. Each record was repeated 10 times to pick up possible variation in usage in the different landscape classes (required later in the calculation).

Water bodies

It was not possible to use site-specific information on the size of stream segment in national-level risk mapping. Beyond any practical considerations, the resolution of the cropping and usage data was insufficient to link a particular crop or pesticide to a particular stream segment. It was suggested that site selection would target small water bodies for monitoring, so the size of the stream segment should be set constant in different locations. However, this would fail to account for the fact that water bodies tend to have larger volumes per unit length in flatter landscapes. Therefore, the median dimensions of streams in different landscapes (Brown *et al*., 2006) were used to calculate concentrations of pesticide in water from a given input via spray drift or drain flow. This approach ignored any reduction in risk for smaller water bodies with faster flow and thus shorter exposure times. The values used in the calculations are given in Table 2.3. The last column of the table compares the volume per unit length of stream in a particular landscape with that of the standard ditch used in regulatory risk assessment (one metre wide, 0.3 m deep; FOCUS, 2002). Average dimensions of streams give smaller volumes per unit length than the FOCUS default in four of the ten landscapes.

Table 2.3: Stream segment characteristics used for risk calculations

Landscape	Description	Width of water (m)	Depth of water (m)	Volume per metre length (L)	Volume per metre length as % of 'FOCUS' ditch
1	River terraces	3.5	0.25	875	292
2	Fenlands	3.0	0.30	900	300
3	Sandlands	2.3	0.15	345	115
4	Eutrophic tills	2.0	0.10	200	67
5	Oligotrophic tills	2.0	0.10	200	67
6	Clays	2.0	0.10	200	67
7	Chalk and limestone	2.25	0.15	338	113
8	Loams	1.6	0.10	160	53
9	Hard rock and clay	3.0	0.10	300	100
10	Hard rock	2.8	0.15	420	140

Exposure calculations

Any attempt to predict exposure via surface run-off would be highly speculative. Thus, exposure calculations only considered entry to water via spray drift and drainage. Spray drift was calculated for all crops. Drainage was only calculated for those combinations of crop and landscape where a significant proportion of the land is drained, as estimated from landscape characteristics (Brown *et al*., 2006) and reference to the SEISMIC database (Hallett *et al*., 1995). Drainage calculations were further restricted so that only applications made between September and April (immediately before or during the main drainage period; Jones and Thomasson, 1985) were taken into the calculation (timing of application was available from the usage statistics).

Spray drift was calculated using the standard drift curve methodology from risk assessment (Rautmann *et al*., 2001). The 90th percentile was used as in regulatory practice, so the output from the calculations reflected the potential risk. Spray drift input was integrated over the width of the stream segment as recommended by FOCUS (2002). No-spray buffer zones were included where imposed, but there was no consideration of the true distance between field and water (characterisation of buffer margins was the major focus of landscape-level analysis later in the project).

A worst-case calculation is used in UK regulatory assessments to calculate predicted environmental concentrations in surface water from drain flow. This calculation is based on an analysis of pesticide transport to drains from the very heavy cracking clay soil present at Brimstone Farm. Here, the approach was extended and refined to make it soil-specific. The method was similar to regulatory practice, but values for loading to water were calculated for a wider range of soils according to the method presented by Brown *et al*. (2003). These broad soil types were correlated with landscape types. Losses in relation to soil type and sorption capacity of the pesticide are given in Table 2.4.

Table 2.4: Predicted losses of pesticide in 10 mm drain flow according to soil type and sorption category (all values are % of pesticide in soil at application)

Soil texture group	Koc category (ml g^{-1})

	<15	15-74	75-499	500-999	1,000-3,999	>4,000
Sands	0	0	0	0	0	0
Light loams	0.0012	0	0	0	0	0
Light silts	0.0026	0.0014	0.0001	0	0	0
Medium loams						
- no moles	0.088	0.078	0.066	0.051	0	0
- with moles	0.054	0.090	0.12	0.0064	0	0
Medium silts	0.34	0.47	0.37	0.15	0	0
Clays	1.9	1.9	0.70	0.50	0.020	0.0080

Exposure in surface water via spray drift was calculated by dividing the loading into the volume of water. For drain flow, the loading was divided into a volume equal to the volume of the stream segment plus the volume equivalent to 10 mm of drain flow from a one hectare field entering a stream segment 100 m in length.

Risk calculations

Three sets of risk calculation were carried out. The first two sets considered potential risk associated with exposure via spray drift and drain flow, respectively. The third set compared exposure via spray drift and drain flow for each combination of pesticide, crop and landscape and carried the larger of the two values into the risk calculation. In every case, risk was normalised for toxicity to either *Daphnia magna* or algae. The LC50 values for *Daphnia* and algae for the respective pesticide were taken from the ecotoxicity database and the toxic loading in water was expressed on the basis of toxic units:

$$TU_{D.magna} = \frac{C_i}{LC50_i}$$

where $TU_{D.magna}$ is the toxic units normalised to the toxicity for *Daphnia magna*, C_i is the concentration in water arising from a particular application and $LC50_i$ is the toxicity of the respective pesticide to *Daphnia magna*. The same approach was used to calculate toxic units normalised for toxicity to algae.

Removal of compounds withdrawn under the EU review programme

Pesticide usage information is collected by the Central Science Laboratory according to a rolling programme for individual crop types. The data available to support national-level risk mapping were collected between 2000 and 2003 according to crop type. Risk calculations showed that some of the largest potential risks were predicted for compounds that were withdrawn in the intervening period (either withdrawn from sale in the UK or in a phase-out period). These compounds were identified (Table 2.5) and a second set of risk calculations and maps were prepared excluding the risk from these compounds. It was not possible to identify the usage of new or existing compounds which replaced the withdrawn ones. The second set of maps was thus based on an approximation of current usage. Risk calculations showed that a few compounds with particularly high ecotoxicity dominate risk for a particular situation, so this shortcoming in the second set of maps is not considered to be a fatal flaw.

Table 2.5: Active ingredients withdrawn from sale since collection of pesticide usage data

Aldicarb	Amitraz	Anthracene oil

Atrazine	Aziprotyne	Benomyl
Bromacil	Carbaryl	Chlorfenvinphos
Cyanazine	Desmetryn	Dichlofluanid
Dicofol	Endosulfan	Fenitrothion
Fenpropathrin	Fentin acetate	Fentin hydroxide
Fluoroglycofen-ethyl	Fomesafen	Gamma-HCH
Heptenophos	Malathion	Maneb
Mephosfolan	Metoxuron	Monolinuron
Pentanochlor	Phosalone	Pirimiphos-methyl
Prometryn	Pyrazophos	Pyrifenox
Simazine	Sodium monochloroacetate	Tar acids
Tar oil	Terbacil	Terbuthylazine
Terbutryn	Triazophos	Zineb

2.2.2 Catchment-level processing

The area related to each catchment, hydrogeologic landscape class and ward was identified in the Geographical Information System (GIS) using an overlay function. Within each of these intersecting areas, the amount of land cover classified as agriculture was quantified for each of the three LCM 2000 classes representing cropped agricultural land. This amount was compared to the ward-level crop census to estimate the amount of area within each catchment that was crop (for each of the 15 crop categories present in the toxic unit tables). The percentage of the catchment composed of each of 15 crops was calculated, and that ratio was used to determine the toxic unit contribution from each crop (based on the region containing the ward). The total sum of all toxic units in the catchment was then determined. The procedure is outlined in Figure 2.1 and described in detail below.

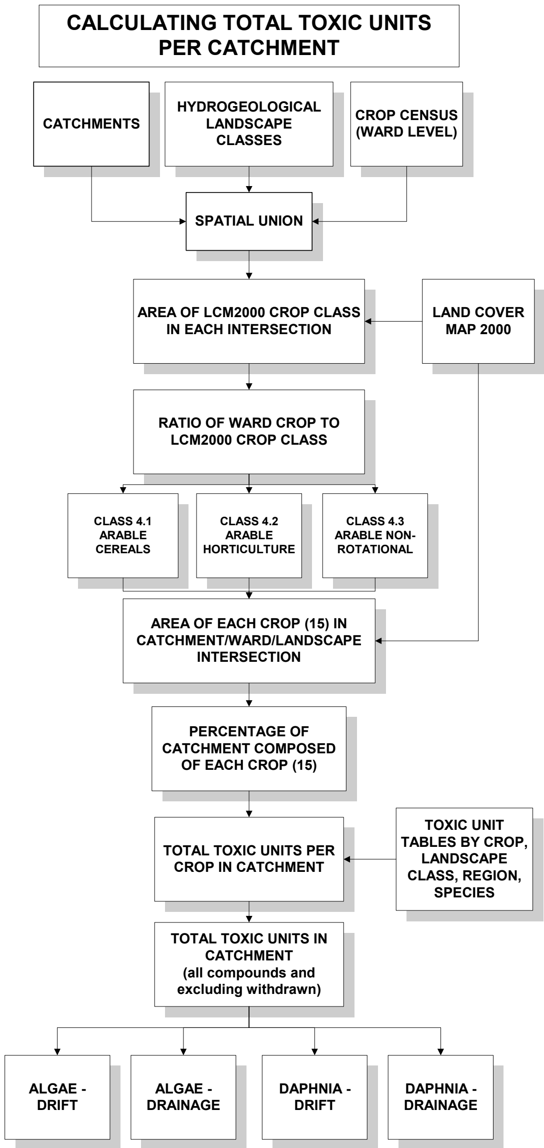


Figure 2.1: Processing flow for calculating summed toxic unit values per catchment

Processing steps

1. Toxic units were generated by aquatic landscape for all applications within the pesticide usage dataset and for a refined set that excluded compounds withdrawn from sale subsequent to collection of usage data. Within each aquatic landscape summary table, toxic units were broken down by organism (algae vs. *Daphnia*), exposure mechanism (drift vs. drainage), region (Northern, Eastern, South-Eastern, South-Western, Midlands and Western, Wales), and crop. Toxic units were given for 15 crop categories (see Table 2.2).
2. Toxic units were appended into one single table with a single toxic unit value given for each combination of landscape, exposure mechanism, organism, crop, and region.
3. Catchments, aquatic landscapes, and ward-level cropping data were spatially combined (union) into one single layer (Figure 2.2).

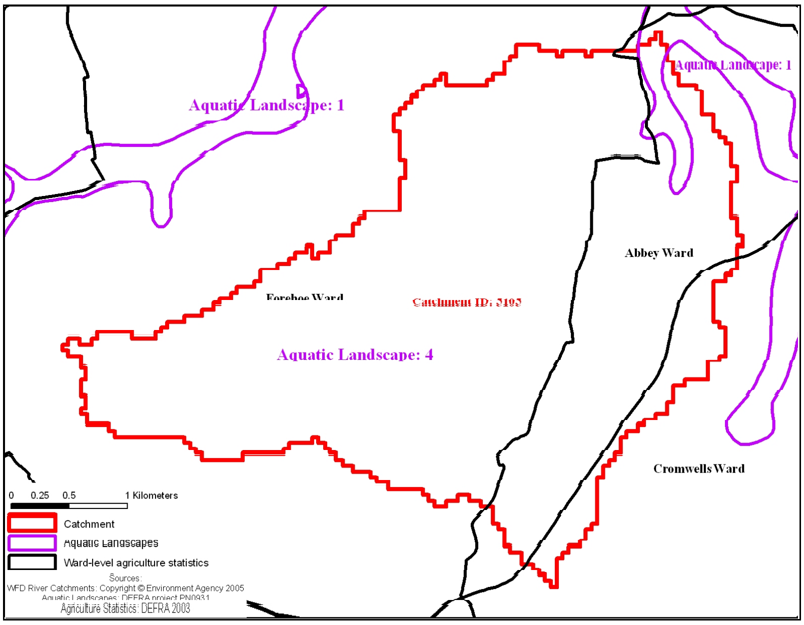


Figure 2.2: Union of catchment, aquatic landscape and ward data

4. LCM 2000 classes were quantified for each polygon generated in Step 3. The classes were selected as all those representing cropped agricultural land (Figure 2.3):

   - 4.1 (arable cereals);
   - 4.2 (arable horticulture)'
   - 4.3 (arable non-rotational).

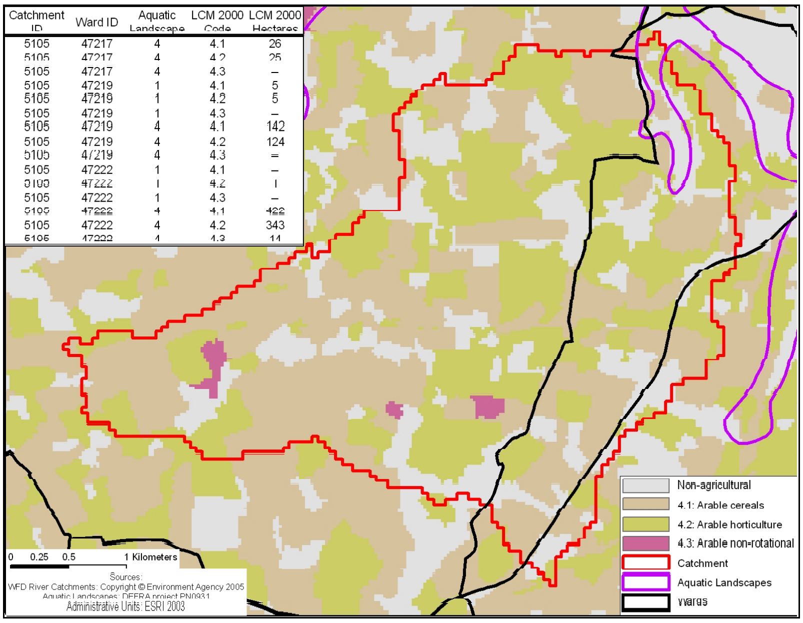


Figure 2.3: Quantification of Land Cover Map 2000 agricultural classes within a catchment

5. The ratio of ward-level crop area (Defra statistics) to appropriate LCM 2000 class quantified by ward in Step 4 was calculated. This ratio represents the percentage of LCM

2000 land cover in the ward that is the crop of interest. As an example, Ward ID 44222 has 279 hectares of LCM 2000 Class 4.1 (arable cereals), and grew 177 hectares of total cereals taken from the ward-level agriculture census. The cropping ratio for total cereals in Ward 44222 is: 177/279 or 63 per cent. In other words, for any portion of the ward, it is assumed that 63 per cent of the LCM 2000 area (Class 4.1) represents cereals.

6. The crop ratio from Step 5 was multiplied by the area of each appropriate LCM 2000 class quantified in each polygon generated from the union of catchments, aquatic landscapes, and wards in Step 3. If the agriculture census reported crop grown in the ward but the appropriate LCM 2000 class was not present, the entire crop area was assumed to be grown in the catchment.
7. Estimated crop areas for each crop taken from Step 6 were summarised by catchment and aquatic landscape.
8. The percentage of each crop area to catchment area was calculated, linking in the region identifier.
9. All estimated percentages from Step 8 were appended into one table.
10. Estimated percentages of each crop at catchment level taken from Step 9 were linked to the toxic units table generated in Step 2. Tables were linked on aquatic landscape, region and crop. Toxic units at the catchment level were calculated as toxic unit values from the toxic units table multiplied by crop percentage of the catchment.
11. Toxic units from Step 10 were summarised by catchment for:

    - spray drift potentially affecting algae;
    - spray drift potentially affecting *Daphnia*;
    - drainage potentially affecting algae;
    - drainage potentially affecting *Daphnia*.

12. Tables generated in Step 11 were joined to the catchment polygon spatial dataset. Maps were generated to show the range of potential toxicity to algae and *Daphnia* by catchment from both spray drift and drainage exposure mechanisms. A set of maps was generated for toxic units calculated for all applications within the pesticide usage dataset and for the refined set that excluded compounds withdrawn from sale subsequent to collection of usage data.

An advantage of the method used to determine crop area for each catchment in Steps 3 to 7 is that the total crop area from the agricultural census is accounted for. This approach used the reported area values from the agricultural census and distributed this area to each catchment based on the land cover data (LCM 2000). If the total area for a specific crop was summed for all catchments in England, it would equate to the total area summed for all wards from the agricultural census. Table 2.6 compares the summed catchment totals to the source agricultural census.

Note that statistics for other vegetables, peas/beans (veg), and soft fruit were only available at the NUTS4 level (local administrative unit, such as a large town or small city), and not at the NUTS5 level (ward). In these cases, the NUTS4 crop areas were used in a similar manner as the ward data, with resulting loss in resolution at the catchment level. Data on hops were also unavailable at the ward level and were estimated in a separate processing step.

Table 2.6: Comparison between cropped areas based on the sum of crop in all catchments and that present in source agricultural census data

Crop	Sum of crop areas from catchment cropping estimates (hectares)	Sum of crop areas from Defra cropping statistics (hectares)	Catchment estimate for area as a % of census area
Beans (field)	114,319	114,319	100%
Bulbs	1,686	1,686	100%
Cereals	927,089	927,092	100%
Hardy nursery stock	284	284	100%
Hops	2,302	2,336	99%
Maize	75,141	75,141	100%
Oilseeds	337,101	337,101	100%
Peas and beans (veg)	9,344	9,354	100%
Peas (dry)	68,731	69,065	100%
Potatoes	38,395	38,521	100%
Other stockfeed	33,685	33,685	100%
Other vegetables	75,193	75,193	100%
Soft fruit	5,774	5,876	98%
Sugar beet	140,114	140,114	100%
Top fruit	16,197	16,197	100%

2.3 Landscape analysis for individual stream segments

2.3.1 Site selection process

Results of national mapping are presented in Section 3.1. The mapping identified orchards as the crop that presents the greatest potential risk to the aquatic environment, so landscape analysis focused on this crop. Areas in or near the counties of Herefordshire, Kent and East Anglia were identified as three important orchard-growing regions in England, each having its distinctive agricultural landscape. Selected areas from within each of these regions were identified for more detailed analysis of orchard proximity and composition of buffers around streams.

The landscape analysis focused on surface water that ran within 20 m of orchards. This distance was selected because spray drift was expected to be the most significant route of exposure for surface waters adjacent to orchards. Exposure via spray drift decreases rapidly as the distance between sprayed area and water increases. For example, drift at three metres and 20 m from the sprayed area of an orchard in late development is calculated to be 15.7 and 1.1 per cent, respectively of the field rate in first-tier risk assessment for pesticides (Rautmann *et al.*, 2001). A systematic approach was used to select surface water within 20 m of orchards in the three key orchard-growing regions. The following steps describe the selection process:

1. A 5 x 5 km grid was developed to overlay the study areas, with each grid cell aligned to OS MasterMap® imagery frames.
2. All inland water was extracted from the OS MasterMap® polygon layer based upon the DESCGROUP field having the ‘inland water’ attribute.

3. All orchards were extracted from the OS MasterMap® polygon layer based upon the DESCGROUP field having the ‘orchards’ attribute or any variation or inclusion of orchards in the attribute description.
4. All OS MasterMap® orchards within 20 m of OS MasterMap® inland water were selected and exported to a new layer. In other words, if an orchard field had any portion located within 20m of water, the entire orchard field was exported (and subsequently summarised in Step 6).
5. The total area of orchard having any portion within 20 m of surface water was calculated for each 5-km grid cell.
6. The 5-km grid cells were then ranked by orchard area near streams.
7. The five 5-km grid cells with the greatest amount of orchard area near streams in each area were selected for water body-level processing. This yielded 125 km^2 in total to be analysed in each of the three areas.

Figure 2.4 shows the 5-km grid cells selected for each region.

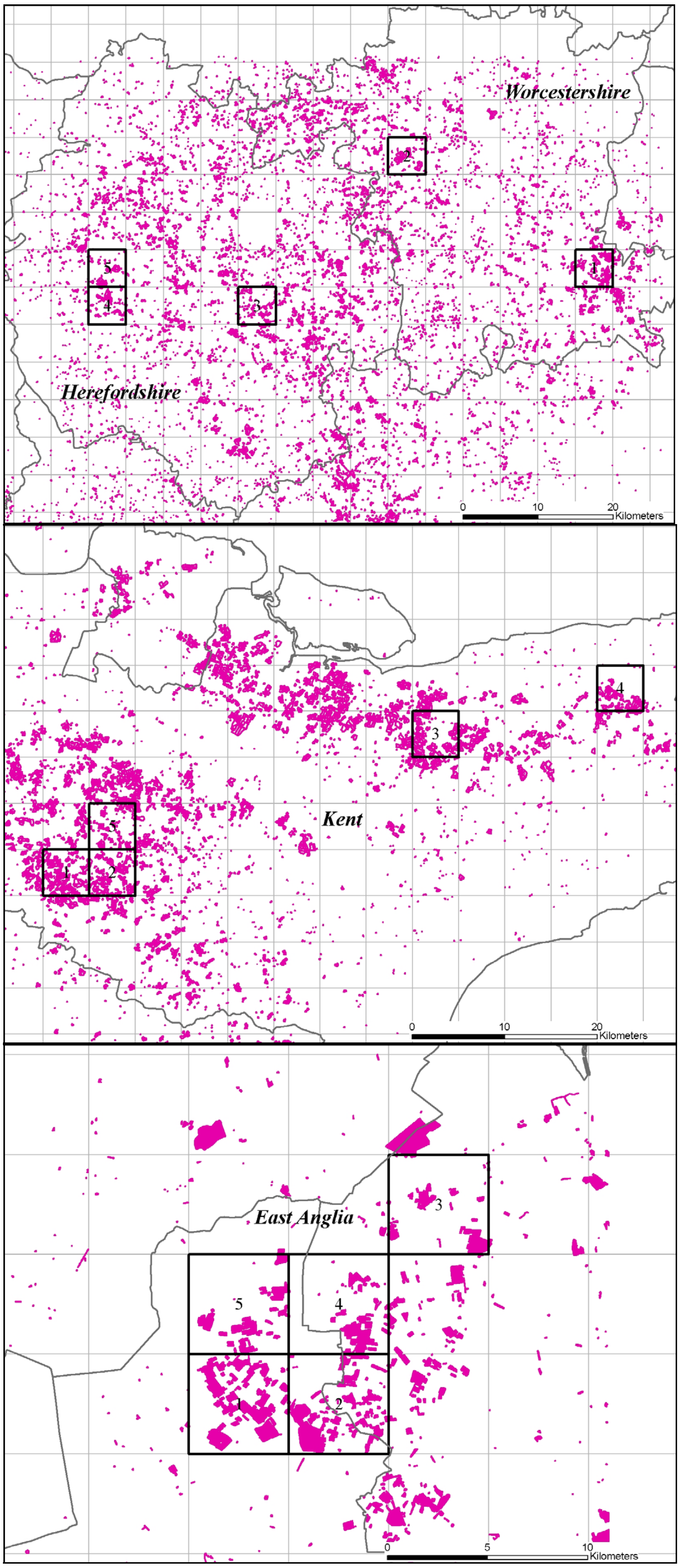


Figure 2.4: Selected 5-km grid cells for Herefordshire, Kent and East Anglia. Gridlines delineate 5 x 5 km cells; shaded areas are orchards; scale varies for the three maps.

2.3.2 Spatial processing – base data layers

To generate detailed information on the proximity of orchards to surface water, and the composition of the intervening buffer, several base data layers were needed for the GIS processing. For each of the fifteen 5-km grid cells selected (top five from each area), information on refined stream location and land cover were generated by an image analyst using OS MasterMap® aerial imagery.

OS MasterMap® polygon land cover classification data and OS MasterMap® aerial imagery (colour, 35-cm resolution) are closely georeferenced, but are not perfectly registered to each other. Since this study aimed to characterise the composition of buffers around potentially vulnerable water, it was important to generate as close a relationship as possible between water and the land cover surrounding it. Both the water and land cover were therefore digitised from the aerial imagery. Streams were first digitised, followed by proximate land cover (areas within 20 m of streams). The process of identifying hydrology and land cover data is described in the following sections.

Hydrology

The spatial hydrology layer generated from the OS MasterMap® data in Step 6 of Section 2.3.1 was used as a guide in digitising water from the aerial imagery. This was particularly important in areas in which the aerial imagery did not clearly demonstrate the existence of surface water. The hydrology layer from Step 6 also served to indicate where the water to be digitised was to begin and end, as work was only undertaken on water within the 20 metre distance to orchards.

Some inconsistencies existed between the edition of the OS MasterMap® polygon land cover classification layer and the aerial imagery, and there were instances in which hydrology extracted from the OS MasterMap® was not actually near orchards at the time of aerial imagery acquisition (due to changes in cropping). Conversely, there were instances in which orchards existed in the imagery, but hydrology was not included in the OS MasterMap® extraction indicating 20 metre proximity to orchards. Surface water was removed from the digitised hydrology dataset in the former example, and added to the hydrology in the latter example.

Water was digitised as a continuous line, even where it flowed under roads, tracks, bridges or vegetation. Where dense trees concealed any indication of stream location, it was assumed that the stream flowed through the middle of the trees. Water having no clearly defined banks on the aerial imagery was digitised as a line, and water having clearly visible banks was digitised as a polygon. Rivers and ponds were not the focus for this study and were excluded from the water classification. However, streams, canals or ditches of less than 8.25 m width were included. The definition of streams as having width < 8.25 m and rivers as having width ≥ 8.25 m is operational and marks the distinction between single and double blue lines on the 1:25,000 Ordnance Survey maps. Hydrology from the OS MasterMap® did not include attributes about water body class (such as stream, canal, ditch), width or permanence, and water digitised from the aerial imagery was not attributed accordingly. Since a size class attribute was absent from the baseline OS MasterMap® hydrology, the decision was made to include any individual stream segment based on the 8.25 m constraint using the measurement tool in ArcMap during the digitising phase.

Land cover

A major objective of the study was to characterise the land cover around streams where orchards existed within 20 m of the stream. Particular attention was given to buffers existing between orchards and streams. Buffers for the streams selected from the three distinct orchard-growing areas were characterised to establish relative vulnerability between the growing regions. Buffers were classified using a visual interpretation of the aerial imagery, and distinct land covers were delineated through on-screen digitising in the GIS. Digitising was performed at a scale of between 1:1,000 and 1:1,500 to ensure sufficient detail in land cover delineations. The following steps were taken for the land cover classification of the buffers:

1. Streams within 20 m of orchards digitised from the aerial imagery were buffered in the GIS by 21 m and exported to a new layer.
2. The 21-m buffers and the digitised hydrology were spatially combined (union operation).
3. The output from Step 2 was overlaid with georeferenced aerial imagery.
4. Land cover was examined and digitised within the 21-m polygons using the aerial imagery.
5. Land cover polygons were assigned one of the land cover class attributes recording the appropriate land cover class.

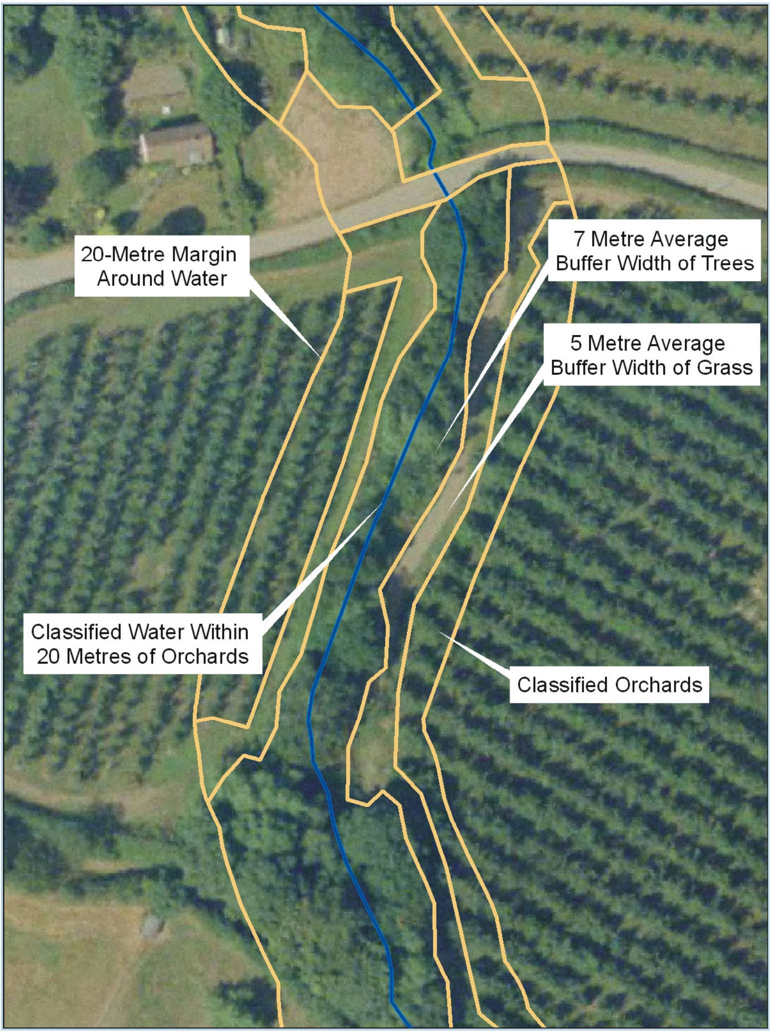


Figure 2.5: Example of digitised hydrology and land cover

Land cover was classified in 20-metre buffers to include orchards and any intervening land cover in the buffer between water and orchards. A distinction between mature and young orchards was made during the classification process, but the two classes were combined later into a single orchards class. Polygons digitised from the imagery were assigned one of the following land cover classifications:

Unclassified	Trees: forest	Arable crop
Other water	Trees: hedgerow	Other crop
Orchards	Scrub	Processed water
Young orchards	Bare ground	
Grass	Urban/developed	

Effort was made to distinguish types of vegetated areas in the buffer. The grass classification was assigned to either mowed or non-maintained grass or pasture, without the presence of tree canopies. The scrub attribute was assigned to rough-textured vegetation, often having a mixture of grass, non-herbaceous woody plants and possibly some small trees, in which any shadows present did not indicate the presence of a mature stand of trees. Trees were identified as either hedgerows or forest. Hedgerows were trees planted in a linear fashion. Trees were attributed as forest if they appeared to be naturally occurring in a non-linear pattern, and not uniformly planted groups of trees. Trees classified as forest were likely to occur near the water, and hedgerows often occurred as breaks between fields.

Non-orchard agriculture was included in the land cover classification. Arable crop was assigned to any row crops, grains or fallow fields. Agricultural land cover that did not appear to be arable crop or orchards was assigned the ‘other crop’ classification. Non-vegetative land cover classes were also identified. Bare ground was considered any non-vegetated land cover without an artificial surface, such as a quarry. The ‘urban/developed’ classification pertained to any roads, paved areas or structures. Water features included in the land cover classification were either stream segments being used in the analysis, classified as ‘processed water’, or water features not evaluated having a width greater than 8.25 metres, classified as ‘other water’.

2.3.3 Presence of surface water and orchards

The total length of hydrology within the five selected 5-km grid cells was calculated using the OS MasterMap® linework hydrology for each area (15 grid cells in total). Grid cells were used to clip the hydrology in order to give a consistent area among sites for comparison purposes. In addition, the total length of hydrology within 20 m of orchards was calculated from the digitised aerial images.

Orchards classified from the aerial imagery included the entire field, even though the orchard was only partially contained within a 20 metre distance of surface water. A summary of orchard area was calculated for the five grid cells selected for each area using the OS MasterMap® polygons attributed as ‘orchards’, or any combination of ‘orchards’ with any other land cover.

2.3.4 Stream perimeter and buffer composition

Water bodies throughout the study area were examined individually for perimeter characteristics and proximity to orchards. To establish a spatial link between the water body and nearby orchards, sampling points were placed at regular intervals along the water body perimeter. Each point represented a specific length of perimeter, that is, if sample points were separated by 10 metre intervals then each point represented 10 m of water body perimeter and it was assumed that values calculated at that point represented the whole 10 m.

Points were placed starting at 2.5 m intervals along the perimeter of streams. In cases where the spatial density of water bodies and limitations within the GIS software caused an overflow condition, the point spacing was increased by 2.5 m for the failing water bodies until all were processed successfully. Specifically, 116 segments failed using the 2.5 m point spacing and were processed using 5 m point spacing. Of the 116 segments processed using 5 m point spacing, three segments failed and were subsequently processed at 10 m point spacing.

Once the sample points were placed, lines were drawn against the potential wind direction in eight compass directions (N, NE, E, SE, S, SW, W and NW) out to 20 m. These lines were intersected with the classified land cover to produce line segments through each individual land cover class. The distance to orchard and the segment lengths through individual land covers were recorded for each stream, point and direction. This information was then used to calculate the perimeter composition, buffer composition and presence of mitigating land cover in the buffer presented in this section. After processing the stream segments in each of the three sites, a total of 316,100 measurements were made from the points along the stream perimeters (Table 2.7).

Table 2.7: Total number of measurements made from perimeter sampling points

Site	Number of stream segments	Total number of measurements
Herefordshire	119	30,636
Kent	354	79,989
East Anglia	1,008	205,475
All sites	1,481	316,100

The composition of the water body perimeter is represented in the idealised water body in Figure 2.6. This example illustrates the three categories of water body perimeter in the basic analysis: direct adjacency, buffered perimeter and non-cropped perimeter. Note that for each perimeter sample point, the direction with the shortest distance to orchard was used for any reporting of adjacency or buffer width. The portion of the perimeter shared with (or within one metre of) orchard was called directly adjacent. The buffered perimeter was defined as that portion of the water body perimeter that had orchard within 20 m, but was not directly adjacent. The non-cropped perimeter had no orchard within 20 m. Only portions of stream segments that were potentially exposed to orchard within 20 m were examined in this study. The number of perimeter sample points in each group was recorded and compared to the total number of perimeter sample points to determine the proportion of the perimeter that fell into each group.

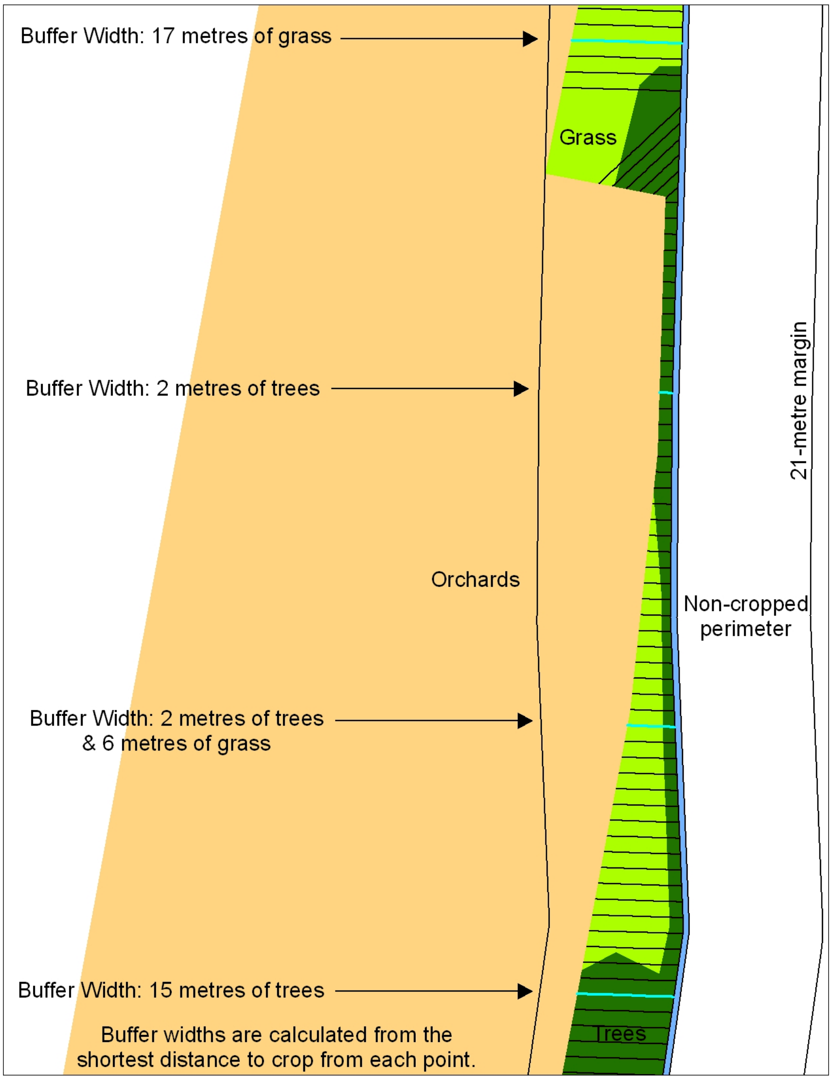


Figure 2.6: Idealised representation of basic water body segment perimeter composition

Figure 2.7 illustrates the extent of the proximate area examined during the spatial processing. In this case the perimeter of the stream was examined every 2.5 m. The figure shows that the areas surrounding water bodies were sufficiently sampled to enable the spatial relationship between orchard and water (direction and distance) to be reliably determined for the entire stream segment.

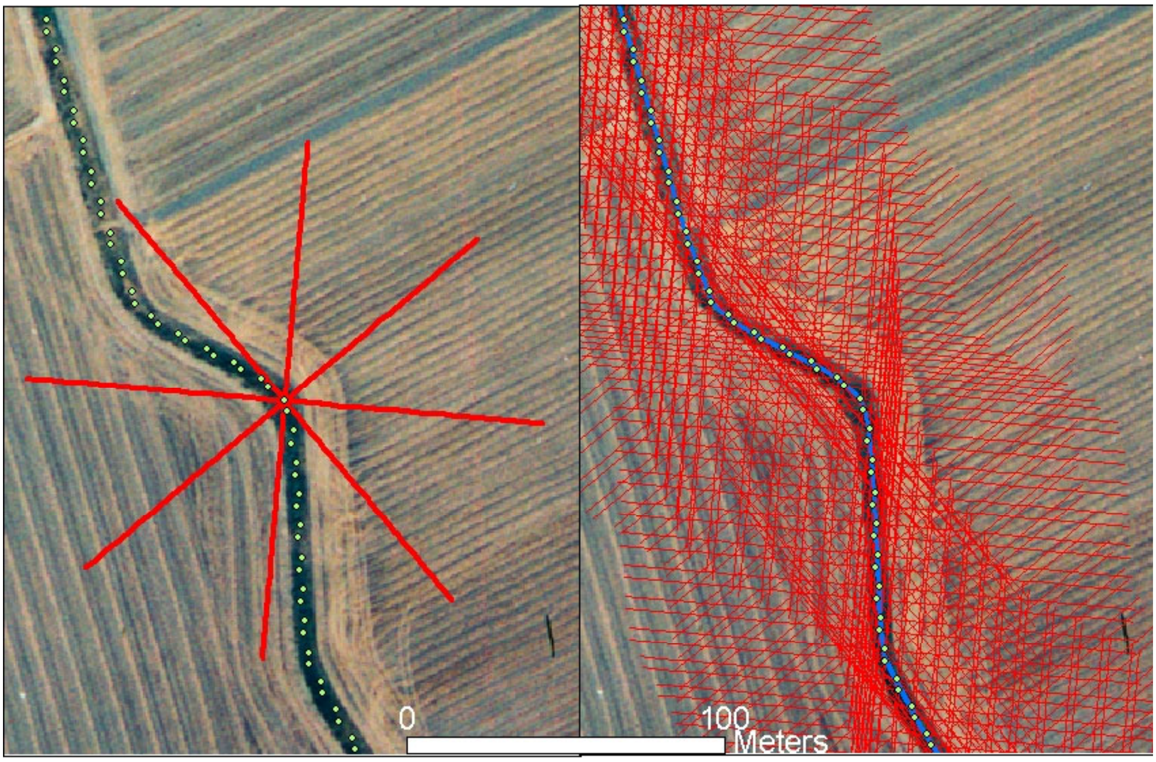


Figure 2.7: Example of sampling density along a stream

The width of the buffer between stream segments and orchards was most important in the perimeter composition, since stream segments were rarely directly adjacent to orchards and never non-cropped due to the nature of the selection and digitising process. Buffer width was summarised by stream segment based on both the shortest distance to orchard and all directions for each point along the stream segment. The average buffer width using both sets of measurements per perimeter sample point was summarised by stream segment. Figure 2.8 shows the numerous direction-related measurements made from a single sample point on a stream and the resulting shortest distance. Note that the direction with the shortest distance to orchard will not always be the same for successive perimeter sample points.

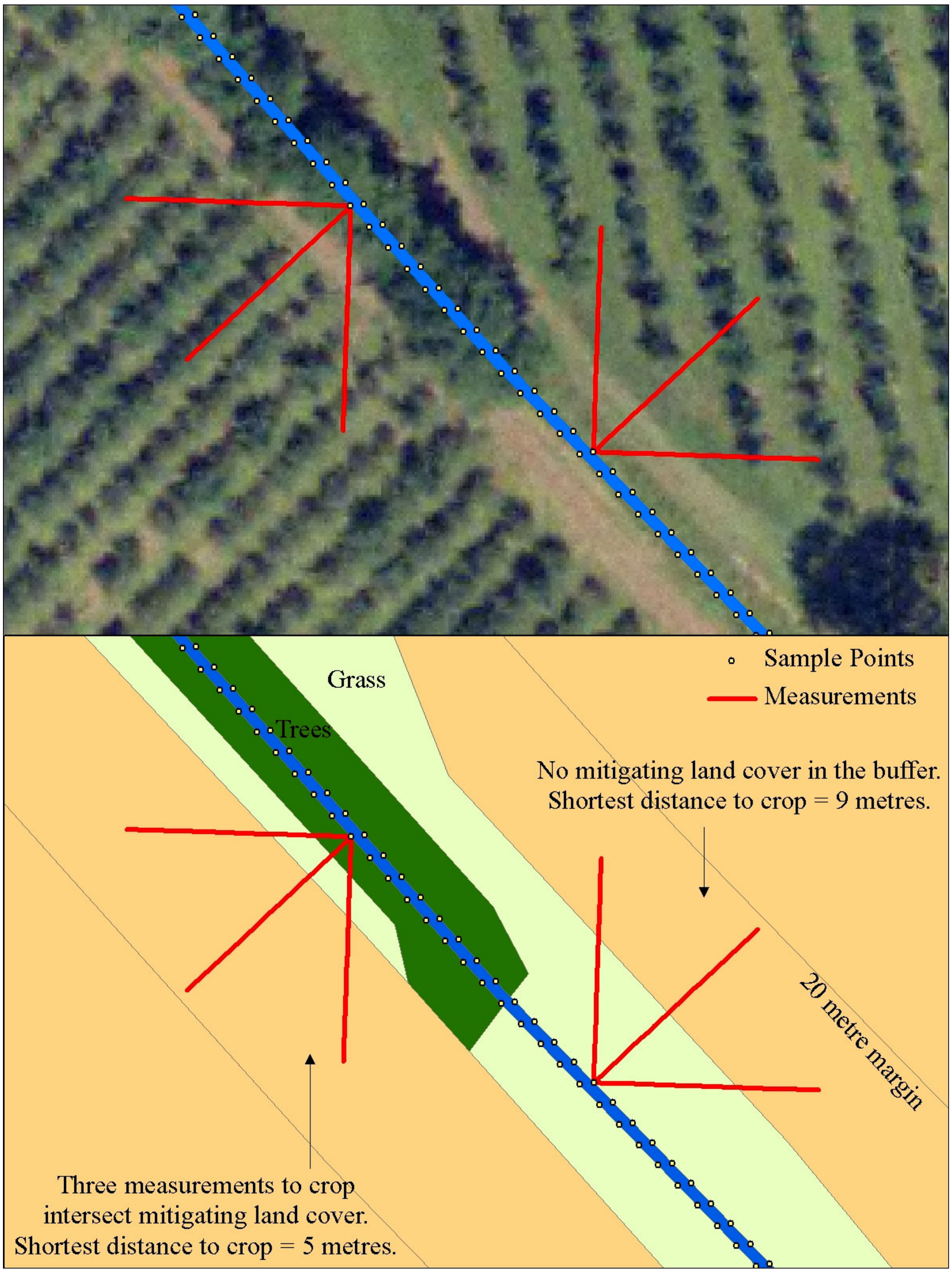


Figure 2.8: Example of selecting the shortest distance to crop for a single sample point (figure is simplified and process was repeated for all nodes on both sides of the water body and for all transects that intercepted crop)

2.3.5 Buffer composition

Results from the analysis of buffer composition show the percentage of buffer area composed of the different land covers when a buffer existed. For the analysis, all directions in which orchard was found for each sample point were used to calculate the composition of the buffer. The segment length through each land cover was recorded for all directions leading to orchards. The total length of segments through each land cover class was then used to determine the percentage of buffer area composed of individual land covers. Figure 2. illustrates how the buffer composition was determined using the distance to orchard for all directions having orchard within

20 m. Using all the directions in which orchard was found offered a greater amount of sampling for the buffer (that is, lengths of land cover passed through to get to orchards).

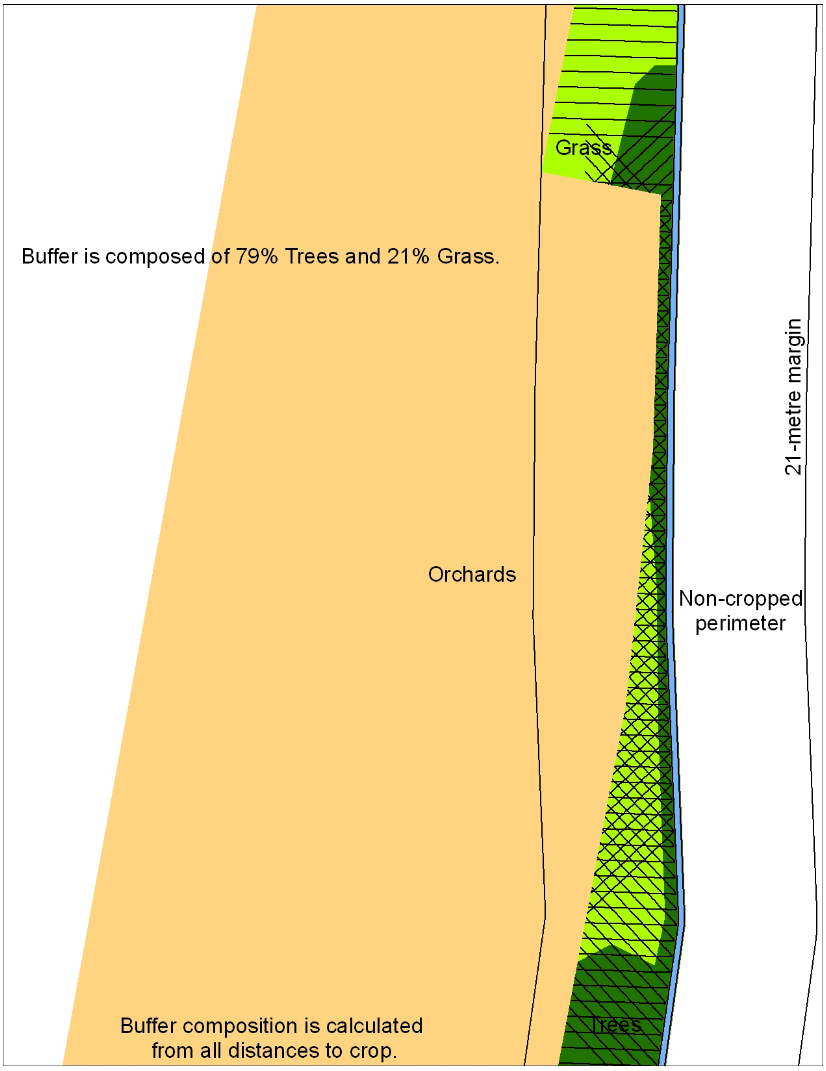


Figure 2.9: Idealised example of the analysis of buffer composition

2.3.6 Presence of mitigating land cover

Buffer composition gives an indication of the types of land cover in the area located between the stream segment perimeter and orchard out to 20 m. Some land covers may provide drift mitigation, so an estimate of the proportion of these potentially mitigating land cover types is important. Mitigating land covers include 'trees' in the form of forests or hedgerows and 'scrub' land cover classes. Note that in this case, the 'forest' class is used when a group of dense trees exist in a non-linear manner. This definition of mitigating vegetation probably underestimates actual mitigation, as rough grassland is excluded but may present a significant barrier to drift. For each sampling transect, if a line segment passed through at least one metre of mitigating land cover the point/direction was considered mitigated. The average width of the mitigating land cover was then calculated for that stream segment using the segment lengths.

2.4 Detailed analysis of the potential risk associated with pesticides used in orchards

Field-level usage data were accessed for 106 orchards on eight farms in Cambridgeshire and Norfolk (loose correspondence to East Anglia). This area was chosen because initial interrogation of aerial photography suggested that the potential for exposure to pesticides was greater in East Anglia than in either Herefordshire or Kent. The crops with usage data were apples and pears, apart from two plum orchards. The data were provisional output from the 2004 orchard usage survey (two years later than the 2002 data used for national risk mapping, where the 2004 data were not available). Risk calculations were repeated at the field scale. The implication of this change in scale was that repeat applications of the same compound on the same field were now represented (and counted twice) and that compounds with high site risk but infrequent use were now properly represented (previously the risk would have been diluted by the spatial averaging process). Exposure was calculated as previously by differentiating between ground sprays and aerial sprays (data shows that only two herbicide applications used a knapsack, all others being machinery ground sprays), and accounting for any statutory no-spray buffer. The total number of spray events (unique combinations of active ingredient and spray date) across the 106 fields was 6,269 (average 59 per field) including repeat applications of the same active ingredient.

2.5 Statistical design of sampling programme

The initial specification for the project indicated that a total of 50 sites should be monitored. Statistical analyses were carried out to provide an indication of the power to detect effects based on this sample size. The study design anticipated that the 50 monitoring sites would be selected to fall within defined classes of toxic loading (e.g. high, medium and low) in one or two environmental scenarios. Power calculations were undertaken to determine the number of replicate sites required within each class to detect an effect of a pre-defined size.

A prerequisite for the statistical analysis was ecological datasets with similar structure (mean counts and variance) to that anticipated within the study. Clearly data are not available to link pesticide concentrations with natural variability in populations of aquatic invertebrates and diatoms in small streams. Instead, aquatic community data were compiled from a number of sources (Table 2.8). These data were collected in water bodies of a variety of sizes and varied significantly in data type, structure of replication and level of taxonomic identification. Power analyses were undertaken for several of the datasets, but the SOWAP dataset was considered the closest analogue to the dataset that would be collected for this project and only this dataset is considered further within this report.

A variety of approaches to power calculation were used to investigate possible influence on the results. These were: simple t-test (one-sided), ANOVA and regression analysis (all undertaken with Genstat v8), and power curve simulation approaches using GLM and based on a log-normal bootstrap, a negative binomial distribution and a Poisson distribution. Discussions within the steering group led to the following criteria for the power calculations using the SOWAP data:

1. It was assumed that the SOWAP assemblages represented conditions in the category of low exposure to pesticides; analysis determined power to detect a specific reduction in species richness (such as a 25 per cent reduction in richness).
2. The analyses targeted 80 per cent power to detect effects as a minimum and 90 per cent as a preferred level.

Table 2.8: Datasets compiled to support statistical design of the sampling programme

Dataset	Description
JNCC river	Species-level macroinvertebrate samples based on three minute netting at 30 sites; sampling in spring, summer and autumn to characterise inter-season variability.
River Cole	Species-level macroinvertebrate samples based on three minute netting; structure allows examination of differences between restored and control river stretches, between years in each of these, between season, between meander and between edge and centre sampling positions.
River Eden	Six diatom indices for six sites at different distances downstream of a sewage treatment works sampled over a period of four to seven years.
River Wear	Statistical analysis of trophic diatom index values upstream and downstream of a sewage treatment works sampled over a period of four to six years.
SOWAP	Species level macroinvertebrate samples from 16 headwater streams in cultivated (11 sites) and semi-natural landscapes (five sites). At each site, 6 x 30 sec samples were collected from a 100 m stream length, each sample being taken from a randomly located 1 m^2 quadrat with a miniature hand net.
Titley Court	Species-level macroinvertebrate data from surber samples taken from three small permanent streams – one adjacent to grassland and two adjacent to winter wheat with either no buffer or a 6-m buffer. Samples collected every four weeks and additionally after pesticide application. Allowed characterisation of temporal variability in abundance and comparison of abundance in the different treatments.
Thames	Species-level macroinvertebrate samples based on three minute netting; samples from a single season at 14 sites one to two km apart to characterise longitudinal variability.

2.6 Site selection and site visits

Site visits were undertaken in three discrete areas to identify potential locations for the monitoring phase of the work. These visits were to Herefordshire, Wisbech in East Anglia, and Kent. The process for undertaking site visits was as follows:

1. National-level risk mapping (see Section 2.2) was used to focus on areas with the greatest potential for exposure to pesticides. This mapping clearly demonstrated that orchards were associated with the largest potential risk and focused attention on Herefordshire, Kent and some smaller areas in East Anglia.

2. GIS analysis was undertaken to identify all orchards within the target locations that were situated within 50 m of a stream (this criterion was selected rather than 20 m so that the number of study locations was not reduced too drastically in this initial step). The datasets used in this analysis were the 1:25,000 water linework from OS MasterMap® and land use data again taken from OS MasterMap®. Orchards identified as adjacent to streams were considered potential sites. More refined analyses identified streams that were within five km of the source and where there was no building with a footprint > 80 m^2 within 100 m of the stream. The purpose of this work was to identify small streams adjacent to orchards that were unlikely to be impacted by soakaways and cesspits from adjacent farms and houses. The latter analyses severely restricted the number of potential sites and were abandoned in favour of detailed site investigations.

3. Aerial photography was overlaid onto water linework and an OS map base in a desk-based analysis of potential sites. The aerial images allowed investigation of conditions around the stream (such as the presence of hedges and woodland between orchard and stream). There was also detailed analysis of anthropogenic influences, location of the stream section within

the broader catchment and land use conditions upstream of the section adjacent to the orchard. Based on this analysis, all sites were graded against the criteria for site selection including likelihood of exposure via spray drift, level of management of the orchard (disused orchards were generally easy to distinguish in aerial imagery) and absence of clear impacts from pollutants other than pesticides (for example, sites with farmyards directly adjacent to an upstream section of the water body were downgraded). All sites were graded as 0 (rejected), 1 (unlikely to be suitable), 2 (some chance of being suitable) or 3 (likely to be suitable).

4. Site visits were undertaken according to Table 2.9. Each set of visits inspected a number of sites, with the most promising sites inspected first. At each potential site, the following records were taken:
    a. The area around the site was walked over and general condition of the stream, orchard and area between the two was noted. Notes included orchards that were marked as organic or clearly had low inputs of pesticides; permanence, accessibility and general condition of the stream; evidence for anthropogenic inputs other than pesticides to the stream; type of vegetation in the buffer between orchard and stream; assessment of the potential for inputs of pesticide to the stream via spray drift (based on adjacency and buffer vegetation that might intercept drift), surface run-off (based on topography and evidence of previous run-off events in the form of rills etc.) and drain flow (based on a search for any evidence of drain outfalls).
    b. A water sample was taken for basic analysis of water chemistry (conductivity, pH, nitrate, phosphate). Additionally, conductivity was measured on site for most of the visits.
    c. The aquatic assemblage present at the site was assessed using a rapid field search method with the objective of characterising fauna (that is, were pesticide sensitive taxa present and if so, were they a significant component of the fauna). At each site, hand netting with kick and sweep sampling was undertaken over a representative 5-10 m stretch of the stream and three to five trays of material were examined to assess the composition of the fauna. At each site total net in water time was around 30 secs with about 15 minutes spent at each site. Dominant taxon groups (identified to species in the field where this was possible, or to higher taxonomic level) were recorded. The first visit to Wisbech on 5 October 2005 concentrated only on aspects a) and b) and a repeat visit to examine biology was undertaken on 31 January 2006.

Table 2.9: Details of site visits

Location	**Herefordshire**	**Wisbech 1**	**Wisbech 2**	**Kent**
Visit date	16/09/2005	05/10/2005	31/01/2006	01/03/2006
Visit team	Colin Brown Jeremy Biggs Pascale Nicolet	Colin Brown	Colin Brown Jeremy Biggs Pascale Nicolet Lorraine Maltby	Colin Brown Mericia Whitfield Robert Aquilina Paul Whitehouse
Number of sites visited	20	20	8	16

3 Results and observations

3.1 National-level risk mapping

The risk to aquatic organisms was calculated for all pesticides applied to specific crops in specific regions and then summed to give a single value for each combination of crop (15 categories), region (6 categories) and landscape class (10 categories). This measure of risk is highly simplified, as the risks from individual applications to a crop across a whole year are summed. Thus, a risk value of one does not imply that the predicted environmental concentration of a specific pesticide equalled the LC50 value on a particular occasion. A risk value of one might arise, for example, if ten pesticides each applied at different times gave predicted environmental concentrations equal to one-tenth of their respective LC50. Examples of toxic units per unit area summed on an annual basis and associated with different crops in different regions are given in Tables 3.1 to 3.4.

Figure 3.1 summarises the calculation of potential toxic units by crop type, showing maximum, minimum and median values. Largest summed toxic units were consistently predicted to occur for *Daphnia magna* and top fruit/hops cultivation. Summed toxic units for these crops were always greater than four. Potential risk to *Daphnia* generally exceeded that to algae for most crops. Cereals were an exception, where risk to algae was generally the larger.

Table 3.1: Summed annual toxicity to *Daphnia magna* calculated as toxic units per unit area based on features of clay landscapes. Crop/region combinations are marked in bold where the summed toxic units have a value greater than one. A zero indicates that the crop is not grown in that region. Some crops may be grown in the region but not in clay landscapes – this is accounted for in subsequent mapping, but cannot be identified from the table.

Crop / Region	E	MW	N	SE	SW	W
Beans	0.43	0.41	0.33	0.43	0.28	0.06
Bulbs	0.49	0.03	0.08	0.15	**9.24**	0
Cereals	0.55	0.41	0.40	0.34	0.32	0.10
Hardy nursery stock	**1.12**	0.67	0.10	**1.91**	0.27	0
Hops	0	**17.0**	0	**14.2**	0	0
Maize	0.05	0.09	0.12	0.11	0.19	0.12
Oilseeds	0.80	**2.45**	0.38	0.51	0.48	0.13
Other stockfeeding	0.28	0.49	0.54	0.12	0.06	**1.36**
Other vegetables	0.84	0.99	0.82	**2.17**	**3.39**	**1.28**
Peas and beans	0.16	0.05	0.05	0.16	0.27	0
Dry peas	0.43	0.4	**10.2**	0.39	0.51	0
Potatoes	0.87	0.77	0.35	0.29	0.45	0.14
Soft fruit	**3.21**	**2.83**	0.53	**2.56**	**1.44**	0.38
Sugar beet	0.10	0.21	0.22	0	0.07	0
Top fruit	**33.0**	**19.2**	0	**19.3**	**22.3**	**33.6**

Table 3.2: Summed annual toxicity to algae calculated as toxic units per unit area based on features of clay landscapes. Crop/region combinations are marked in bold where the summed toxic units have a value greater than one. A zero indicates that the crop is not grown in that region. Some crops may be grown in the region but not in clay landscapes – this is accounted for in subsequent mapping, but cannot be identified from the table.

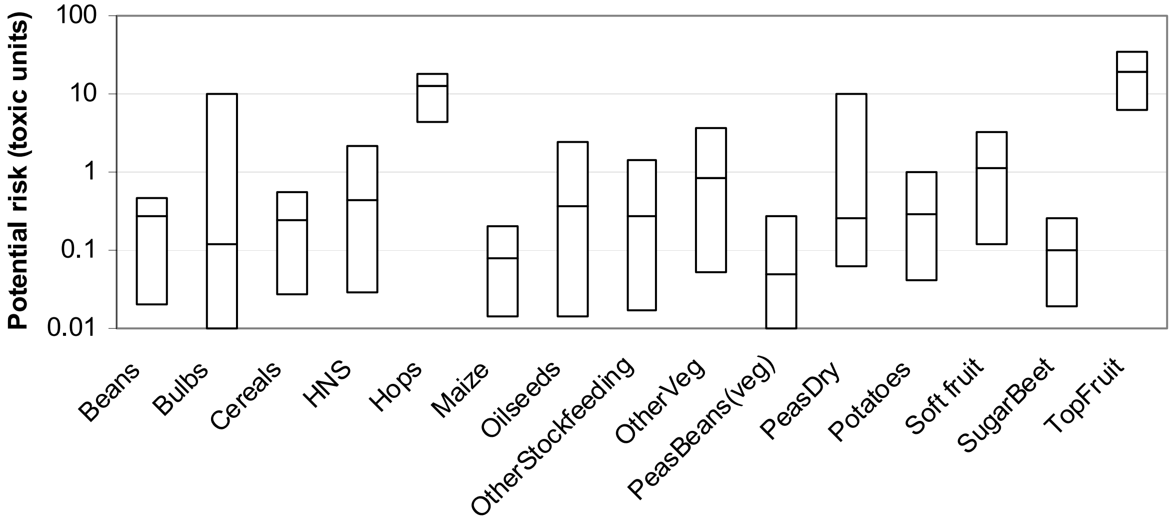
Crop / Region	E	MW	N	SE	SW	W
Beans	0.16	0.17	0.22	0.42	0.19	0.74
Bulbs	0.61	0.88	0.30	0.76	**1.26**	0
Cereals	**2.32**	**2.51**	**3.10**	**2.89**	**2.20**	**1.01**
Hardy nursery stock	**0.91**	0.48	0.58	**1.26**	0.45	0
Hops	0	**7.98**	0	**6.81**	0	0
Maize	0.12	0.10	0.16	0.07	0.17	0.19
Oilseeds	0.95	0.84	0.80	**1.06**	**1.14**	0.01
Other stockfeeding	0.28	0.11	0.28	0.14	0.08	0.05
Other vegetables	**4.67**	**1.52**	**4.31**	**1.61**	0.49	0.06
Peas and beans	0.02	0.13	0.02	0.22	0.01	0
Dry peas	0.34	0.62	0.80	0.68	0.62	0
Potatoes	0.69	0.80	0.59	0.99	0.53	0.24
Soft fruit	0.61	0.30	**1.12**	0.48	0.94	0.40
Sugar beet	0.26	0.26	0.26	0	0.21	0
Top fruit	**6.40**	**13.76**	0	**3.76**	**5.94**	**24.0**

Table 3.3: Summed annual toxicity to *Daphnia magna* calculated as toxic units per unit area based on features of loam landscapes. Crop/region combinations are marked in bold where the summed toxic units have a value greater than one. A zero indicates that the crop is not grown in that region. Some crops may be grown in the region but not in loam landscapes – this is accounted for in subsequent mapping, but cannot be identified from the table.

Crop / Region	E	MW	N	SE	SW	W
Beans	0.45	0.44	0.35	0.45	0.3	0.07
Bulbs	0.53	0.04	0.09	0.17	**9.80**	0
Cereals	0.53	0.39	0.38	0.32	0.31	0.10
Hardy nursery stock	**1.25**	0.74	0.11	**2.14**	0.31	0
Hops	0	**18.4**	0	**15.4**	0	0
Maize	0.06	0.10	0.14	0.12	0.21	0.13
Oilseeds	0.83	0.72	0.39	0.54	0.49	0.06
Other stockfeeding	0.31	0.52	0.57	0.13	0.06	**1.44**
Other vegetables	0.84	**1.09**	0.94	**2.24**	**3.72**	0.20
Peas and beans	0.17	0.04	0.06	0.18	0.16	0
Dry peas	0.45	0.24	**0.66**	0.40	0.51	0
Potatoes	**1.00**	0.85	0.40	0.34	0.52	0.16
Soft fruit	**3.34**	**2.95**	0.56	**2.70**	**1.53**	0.41
Sugar beet	0.12	0.23	0.26	0	0.08	0
Top fruit	**34.3**	**20.1**	0	**20.1**	**23.4**	**35.5**

Table 3.4: Summed annual toxicity to algae calculated as toxic units per unit area based on features of loam landscapes. Crop/region combinations are marked in bold where the summed toxic units have a value greater than one. A zero indicates that the crop is not grown in that region. Some crops may be grown in the region but not in loam landscapes – this is accounted for in subsequent mapping, but cannot be identified from the table.

Crop / Region	E	MW	N	SE	SW	W
Beans	0.17	0.18	0.23	0.45	0.20	0.78
Bulbs	0.68	**1.01**	0.32	0.84	**1.45**	0
Cereals	0.85	0.84	**1.00**	0.99	0.84	0.40
Hardy nursery stock	0.99	0.52	0.63	**1.36**	0.48	0
Hops	0	**8.64**	0	**7.37**	0	0
Maize	0.12	0.10	0.17	0.07	0.18	0.20
Oilseeds	0.07	0.09	0.05	0.08	0.08	0.01
Other stockfeeding	0.31	0.13	0.32	0.15	0.08	0.06
Other vegetables	**1.87**	**1.66**	**1.86**	**1.60**	0.55	0.06
Peas and beans	0.02	0.14	0.03	0.25	0.01	0
Dry peas	0.37	0.67	0.56	0.71	0.54	0
Potatoes	0.77	0.90	0.66	**1.12**	0.58	0.28
Soft fruit	0.65	0.32	**1.19**	0.52	**1.01**	0.43
Sugar beet	0.30	0.30	0.30	0	0.24	0
Top fruit	**6.88**	**14.6**	0	**4.07**	**6.34**	**25.5**



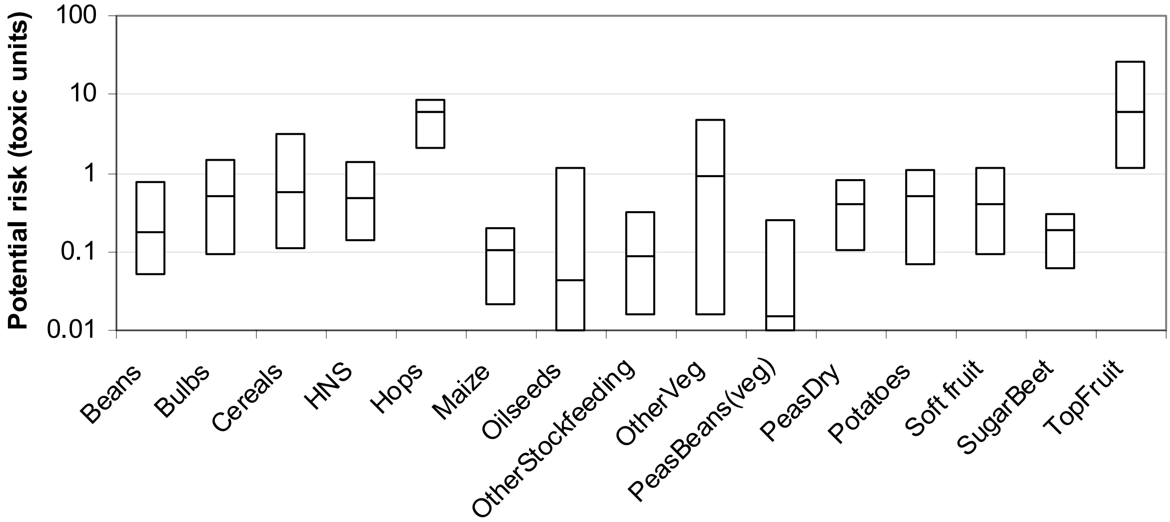


Figure 3.1: Potential risk to *Daphnia magna* (top chart) and algae (bottom chart) calculated for the fifteen crop categories. The lower and upper bounds of the bars show the range for all combinations of landscape type and region (60 values per crop); the line within the bar shows the median value.

Maps of potential (first tier) risk to *Daphnia magna* and algae are shown in Figures 3.2 to 3.4. The first set of maps (Figure 3.2) show the potential risk arising from the largest exposure via either spray drift or drain flow, whilst Figure 3.3 and Figure 3.4 show the potential risk associated with the separate processes of spray drift or drain flow respectively. The division into risk classes is the same in each map and gives roughly equal areas of the top four classes. Each risk map gives an indication of the average toxic loading to streams in each catchment, assuming that streams are adjacent to all land treated with pesticide and that the distance between water and treated land is the same as used in first-tier risk assessment (one metre for arable crops, three metres for orchards and so on, except where mandatory buffers are imposed). No account is taken of the time course of exposure, so: (i) multiple applications of a single active ingredient are assumed to be applied concurrently; (ii) only the largest exposure to a particular active ingredient is considered, even if multiple exposures occur (such as via drain flow); and (iii) the toxic units are summed across the year independent of the times of application.

There are two versions of each map. The first considered pesticide usage data collected between 2000 and 2003. Significant risks were found to be associated with compounds that had been withdrawn in the intervening period (either withdrawn or in a phase-out period). These compounds were identified and a second set of maps was constructed excluding the risk from these compounds. It was not possible to identify the usage of new or existing compounds that had replaced the withdrawn ones. It is thus likely that the first set of maps overestimates the potential risk from current usage and the second set underestimates the risk.

Figure 3.2 shows significant variability in the potential risk within catchments. The risk is averaged across the full area of each catchment, independent of land use, so differences are partly driven by the extent of cultivation of arable crops and fruit. Nevertheless, there are significant differences in summed toxic units for different crops and this shows up in the maps of potential risk. Top fruit, hops and soft fruit have particularly large values for summed toxic units for *Daphnia magna* and this translates into areas with greatest risk in Herefordshire, Kent and in isolated catchments in East Anglia. By comparison, risk is significantly lower in the most intense areas of arable cultivation in the east of England. Risk to algae associated with different crops is less differentiated. Herefordshire and Kent are still the areas associated with greatest risk, but arable areas are also predicted to have significant potential risk for algae, primarily because of high-usage herbicides applied to widespread crops such as cereals and oilseeds.

Previous mapping by the Environment Agency in support of the Water Framework Directive identified areas of high risk to the aquatic environment in the intensive area of arable cultivation around the Wash (www.environment-agency.gov.uk/wfd). Current mapping thus contrasts with the earlier result. One of the key differences is that the mapping undertaken here accounts for characteristics of the landscape. Water bodies in the Fenland landscape around the Wash tend to be significantly wider and deeper than those in most other parts of the country, increasing the potential dilution of any pesticide transported to water.

In most cases, potential risk was dominated by a small number of compounds. The active ingredients that dominated risk for crops with the largest toxic loading are shown in Table 3.5 for an example region and landscape class. *Daphnia magna* is primarily predicted to be affected by insecticides as expected. Herbicides dominate the potential risk to algae for arable crops, whereas insecticides and fungicides are more important for fruit cultivation. Loss of herbicides from orchards was restricted because there are no drains and spray drift is mitigated to some extent by the minimum distance to water of three metres.

Table 3.5: Pesticides that contribute most to the potential risk to *Daphnia magna* and algae associated with different crop groupings. Calculation is shown for crops that have significant cultivation in loam landscapes of the Midland and Western region and also for top fruit only on loam landscapes in the South Eastern region. Pesticides recently withdrawn from sale have been removed. Pesticides shown in bold are predicted to contribute more than 30 per cent of the total potential risk for that crop.

Crop	**Toxic loading to *Daphnia***		**Toxic loading to algae**	
	Total for crop	**Main contributors**	**Total for crop**	**Main contributors**
Cereals	0.39	**Cypermethrin** Chlorpyrifos	0.84	Chlorotoluron Flufenacet **Isoproturon** **Pendimethalin**
Hops	18.45	**Copper oxychloride** Cypermethrin Diquat	8.64	Copper oxychloride **Diquat** Paraquat Pendimethalin Quinoxyfen
Other veg	1.09	Carbosulfan **Chlorpyrifos** Cypermethrin Pirimicarb	1.66	Pendimethalin **Propachlor**
Potatoes	0.85	**Cypermethrin** Diquat Mancozeb Pirimicarb	0.90	D**iquat** Linuron Mancozeb Zoxamide
Top fruit	20.10	**Chlorpyrifos** **Dodine**	14.65	Captan **Dodine** Kresoxim-methl
Top fruit (SE)	20.13	Carbendazim **Chlorpyrifos** Clofentezine Copper oxychloride Cypermethrin Dodine Pirimicarb	4.07	**Captan** Carbendazim Copper oxychloride Diuron **Dodine** Kresoxim-methyl Mancozeb

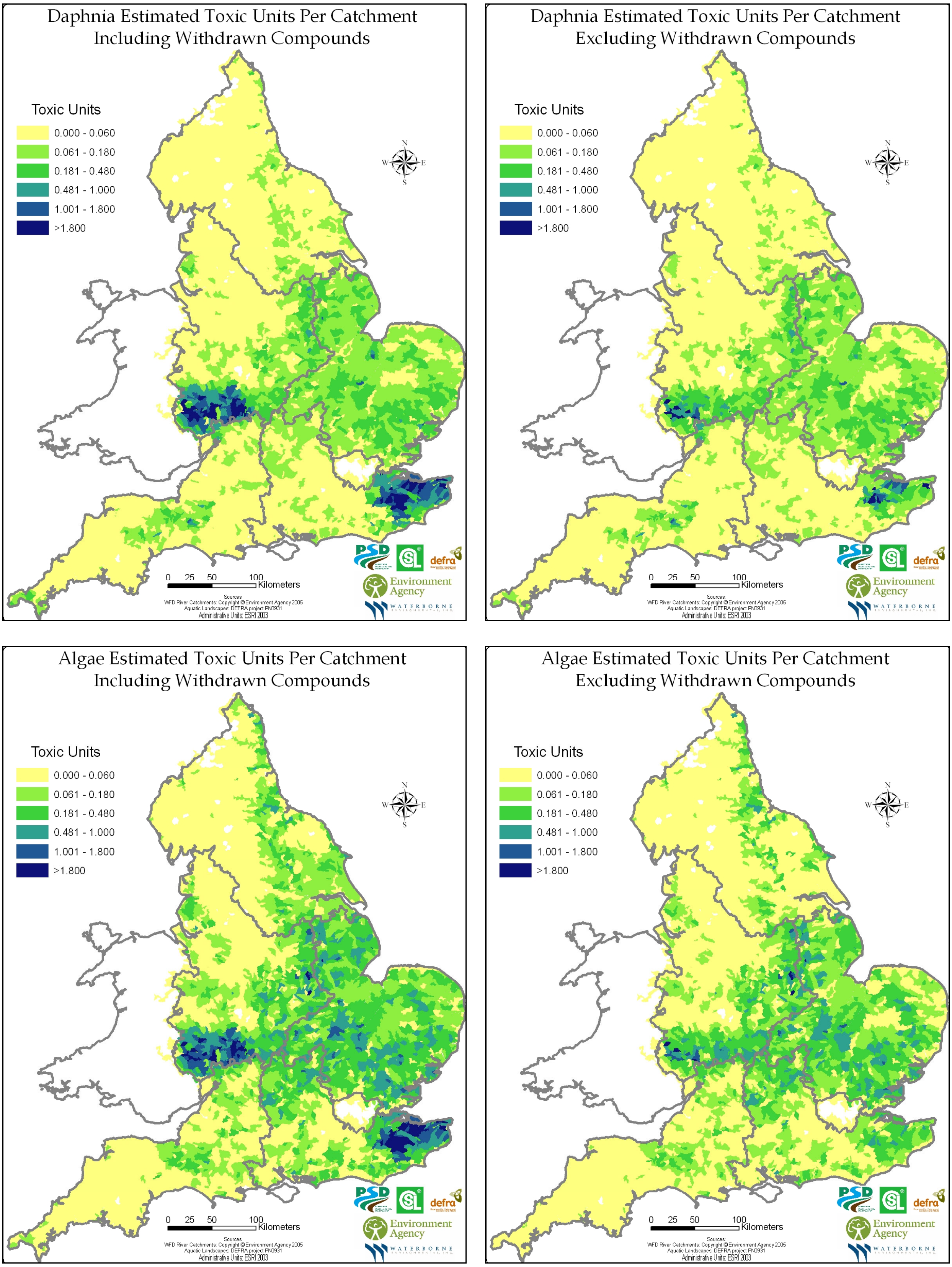


Figure 3.2: Maps showing the potential risk to indicator aquatic organisms (*Daphnia magna* and algae). These are based either on pesticide usage information collected between 2000 and 2003 (according to crop type) or based on the same usage information but with compounds removed that have subsequently had UK registrations revoked.

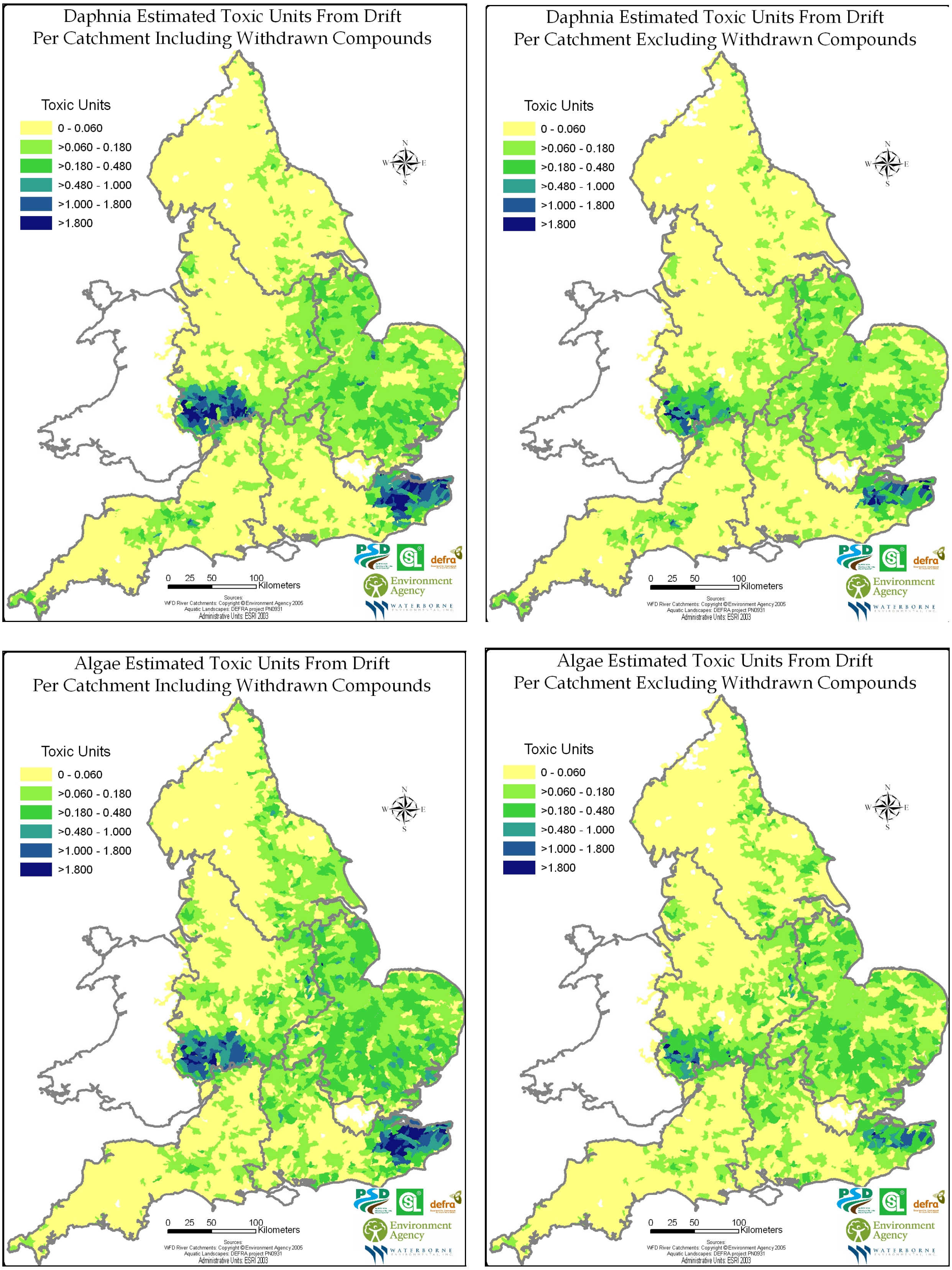


Figure 3.3: Maps showing the potential risk to indicator aquatic organisms (*Daphnia magna* and algae) following exposure to pesticides via spray drift. These are based either on pesticide usage information collected between 2000 and 2003 (according to crop type) or based on the same usage information but with withdrawn compounds removed.

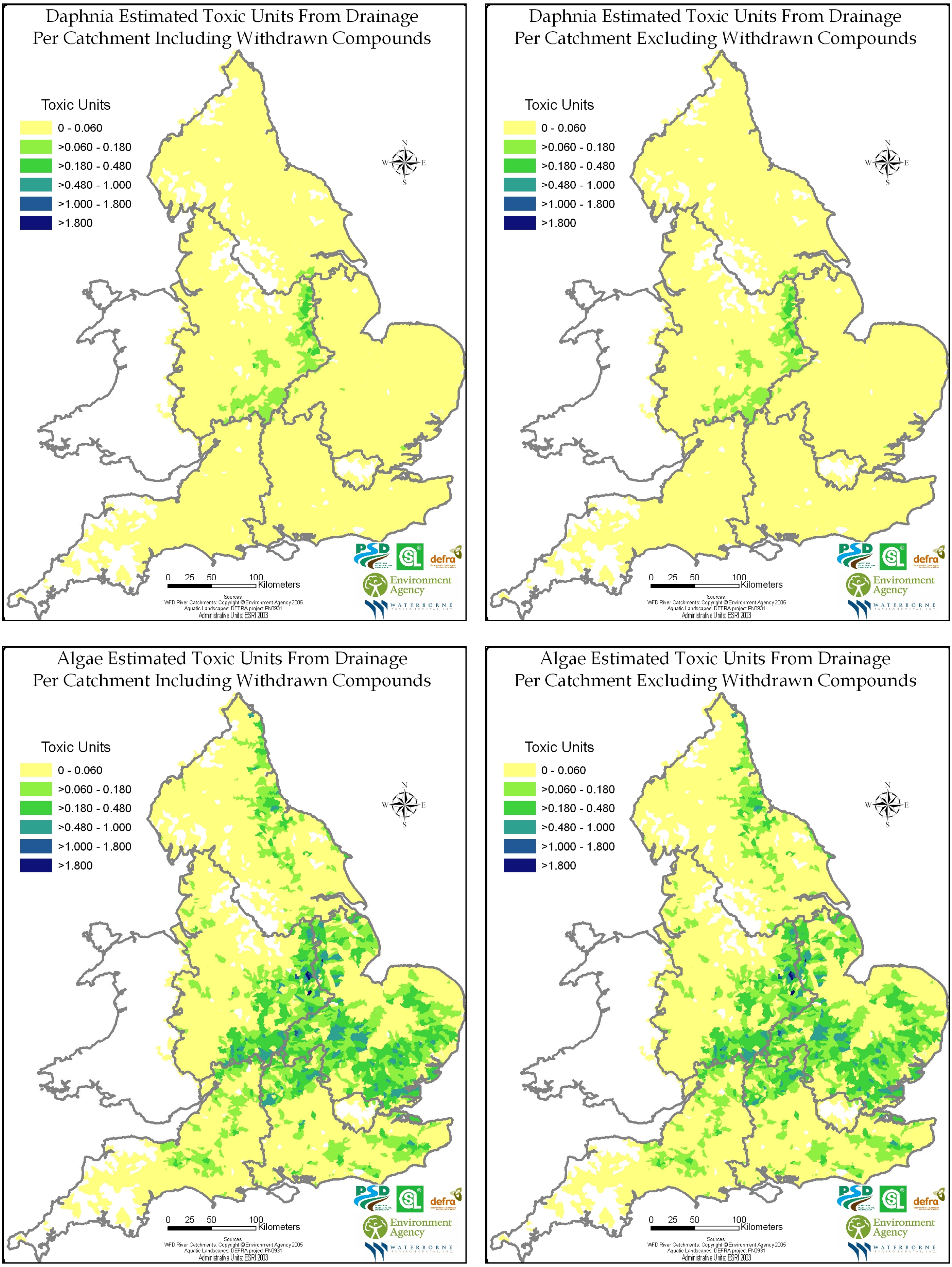
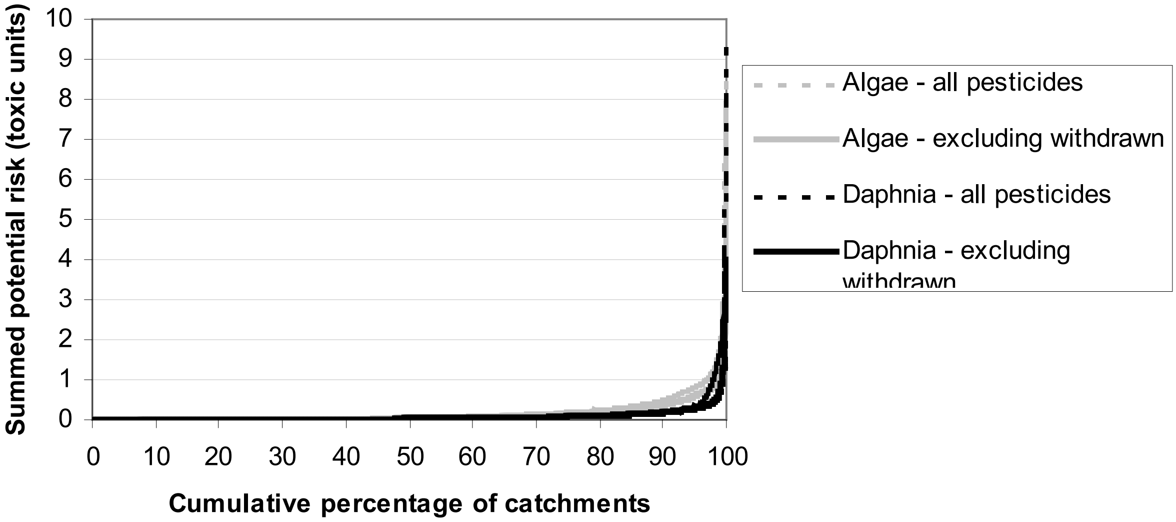


Figure 3.4: Maps showing the potential risk to indicator aquatic organisms (*Daphnia magna* and algae) following exposure to pesticides via drain flow. These are based either on pesticide usage information collected between 2000 and 2003 (according to crop type) or based on the same usage information but with withdrawn compounds removed.

The calculations above identify orchard and hop cultivation as the cropping systems with greatest potential risk and the maps identify Herefordshire and Kent as the areas with greatest intensity of high-risk situations. Data presented in the maps can also be expressed in graphical form to provide context on the distribution of risk situations at the national level. Figure 3.5 plots the ranked distribution of averaged potential risk for all catchments in England. Figure 3.6 presents the same information, but catchment size is also factored in so that the land area with particular categories of averaged potential risk can be determined.

Based on the full pesticide usage dataset, 97 per cent of catchments and 96 per cent of the land area of England is predicted to have summed potential risk averaged at the catchment level of less than one toxic unit per year when normalised to toxicity of either *Daphnia* or algae (again, one toxic unit is selected as an arbitrary threshold). When compounds withdrawn from sale are excluded, 99 per cent of catchments and more than 98 per cent of the land area have predicted risk less than one toxic unit per year. The predicted risk to algae is greater than that to *Daphnia* at this scale of aggregation, except at the extreme tails in the distributions.



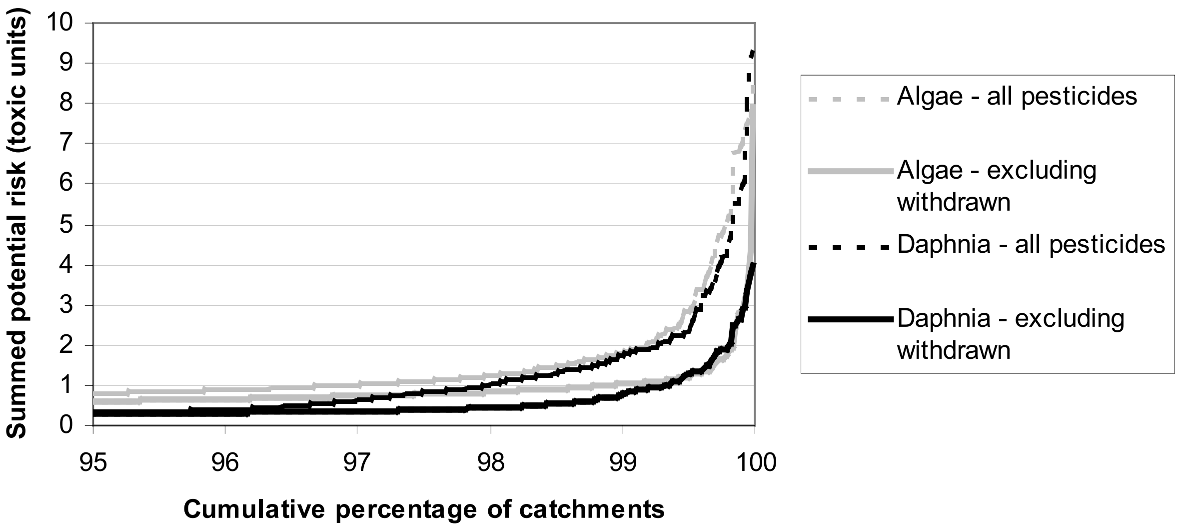
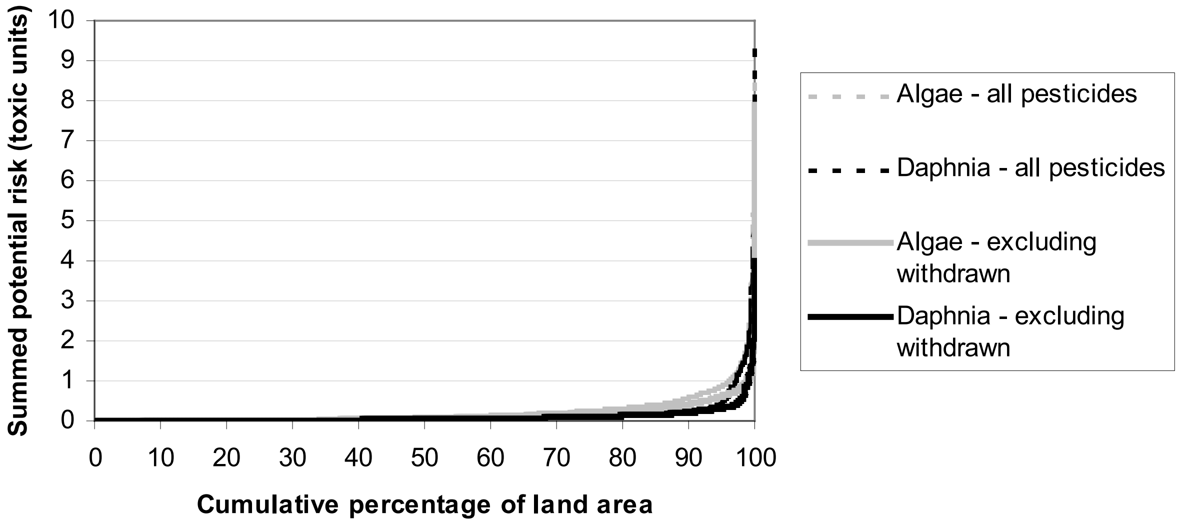


Figure 3.5: Cumulative distributions for the summed toxic units in individual catchments; full distribution (above) and upper five per cent of catchments (below)



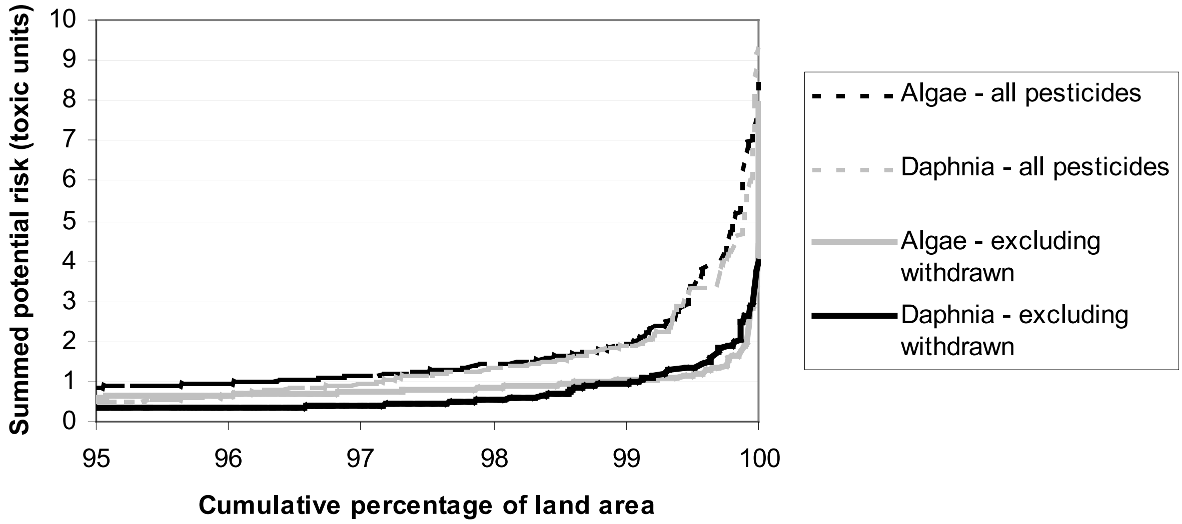


Figure 3.6: Cumulative distributions for the summed toxic units calculated on an area basis for the whole of England; full distribution (above) and upper five per cent of the land area (below)

Simple multiplication of areas for each combination of crop, region and landscape class by the relevant toxic unit loading from risk calculations allowed the risk to be aggregated to national scale. Figure 3.7 shows that the aggregated risk to algae was predicted to be almost 50 per cent larger than that to *Daphnia,* regardless of whether calculations were made with the full pesticide usage dataset from 2000-2003 or without compounds withdrawn in the intervening period. Note that the risk units in Figure 3.7 become entirely arbitrary at this scale.

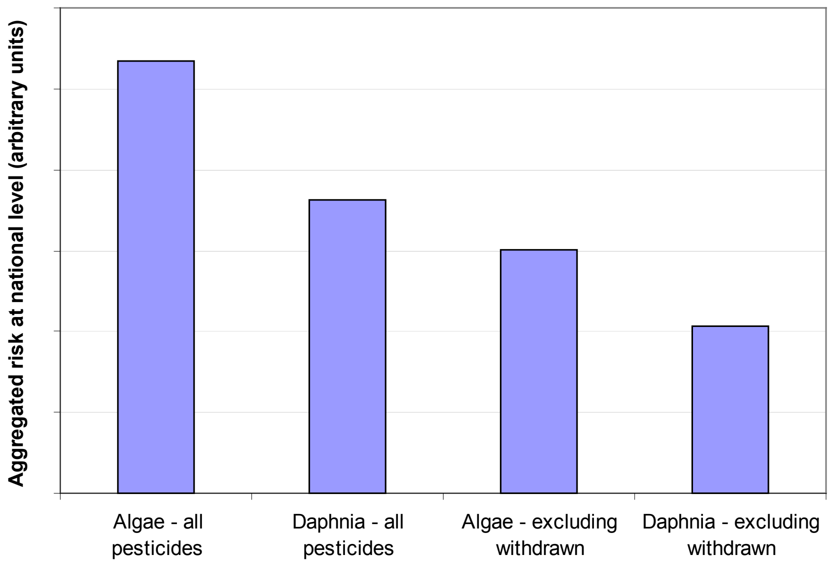


Figure 3.7: Aggregated risk at the national scale for all combinations of crop, region and landscape

Although orchards and hops were predicted to have the greatest potential risk at an individual location, these are relatively minor crops in terms of UK agriculture as a whole. Figure 3.8 shows the relative contribution of different crops to the aggregated risk to adjacent surface waters when expressed across the whole of England. In this calculation, the areas of cultivation for the respective crops become more important for the calculation of relative contribution to risk. Based on the original usage information, cereals are the biggest contributors to risk to algae when averaged nationally, followed by hops and top fruit. When withdrawn compounds are removed, cereals contribute nearly 75 per cent of the risk to algae, while top fruit and oilseeds contribute seven to eight per cent each and the contribution from hops is very small. Hops were the major contributor to risk to *Daphnia* based on original usage data, with top fruit, cereals and oilseed all making significant contributions. Once withdrawn compounds are removed, top fruit and cereals contribute about 30 per cent of the total risk and oilseeds account for a further 20 per cent. The large risk to *Daphnia* from hops arises despite the small area of cultivation and is associated primarily with use of tar oils as defoliants (which has now been discontinued). The predicted contribution of hops to total risk decreased by more than 90 per cent for both indicator species when withdrawn compounds were removed from the calculation. Bulbs, hardy nursery stock and other stockfeeding crops make an insignificant contribution to risk and are not shown.

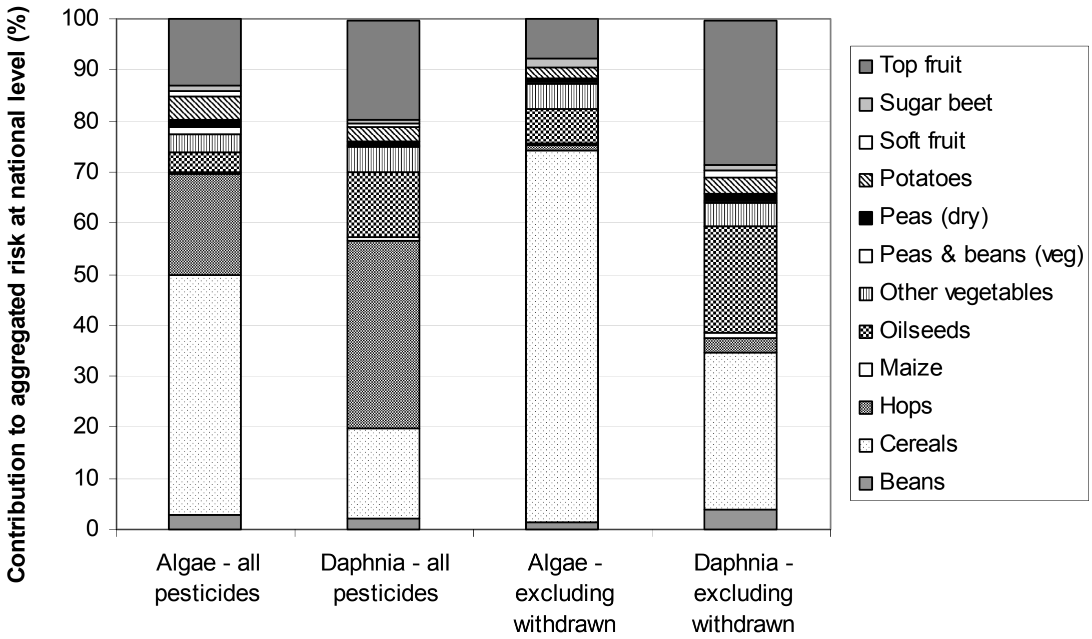


Figure 3.8: Relative contribution of the different crop types to the aggregated risk shown in Figure 3.7

Appendix 2 gives similar breakdowns of the relative contributions of different landscape classes and different regions to the aggregated risk to adjacent surface waters when expressed across the whole of England.

3.2 Landscape-level analysis of areas with intensive orchard cultivation

National risk mapping had clearly identified orchards as the crop associated with the greatest potential risk to adjacent streams. Landscape-level analysis thus focused on this crop and sought to establish the extent to which the conditions leading to risk actually occur in the field. The areas in Herefordshire, Kent and East Anglia that were selected for landscape analysis represent distinct orchard-growing landscapes. While all sites have a relatively high proportion of orchards, the distribution, density, buffer composition and proximity of orchards to water can be quite different. Each area was divided into 5 x 5 km cells and the five cells from each area having the greatest amount of orchard with some portion within 20 m of water were selected for detailed analysis of local conditions.

3.2.1 Presence of surface water

The total length of all streams and the total length, average length and number of stream segments within 20 m of orchards are presented in Table 3.6. East Anglia had both the greatest length of water per unit area and the largest percentage of this water within 20 m of orchards. The lowest value for both of these measures was for Herefordshire, with Kent intermediate in both cases. The Kent grid squares fall into two distinct landscapes. Squares to the south-east of Maidstone are relatively similar to Herefordshire, with lower density of water and a predominance of streams over ditches. Squares on the North Kent coast are much more akin to the Wisbech locations in East Anglia, with dense networks of drainage ditches. Thus, summed measures for Kent tend to fall between the extremes of Herefordshire and East Anglia.

Figure 3.9 shows the range of hydrologic length and density between the three sites. The East Anglia sites had the greatest length of streams, canals and ditches (994 km), followed by the Kent and Herefordshire sites. Examining the variation in hydrologic density within each site (right side of Figure 3.9) shows that Kent had the greatest amount of variability when summarising on the 5-km grid cells. Figure 3.10 presents the distribution of stream segment lengths between the three sites.

Table 3.6: Comparison of digitised hydrology to OS MasterMap® hydrology

Site	Total length of OS MasterMap hydrology (km)	Total length of digitised hydrology within 20 m of Orchards (km)	Digitised hydrology as % of OS MasterMap hydrology	Number of stream segments within 20 m of orchards	Average length of stream segments within 20 m of orchards (m)
Herefordshire	269	29	11%	119	242
Kent	489	68	14%	354	191
East Anglia	994	174	18%	1,008	173

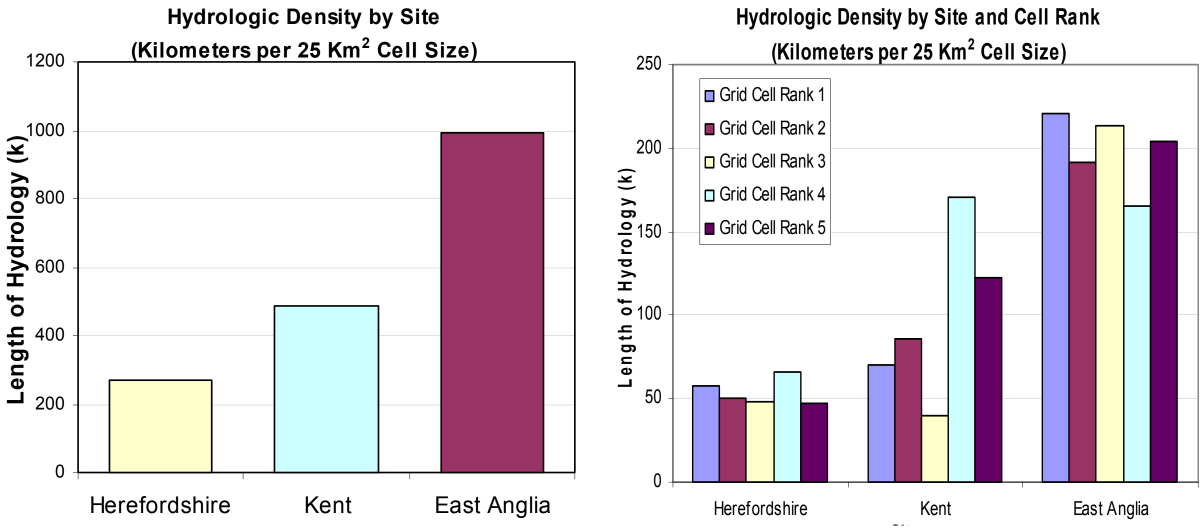


Figure 3.9: Comparison of hydrologic length (left-hand chart) and hydrologic density (right-hand chart) for Herefordshire, Kent and East Anglia

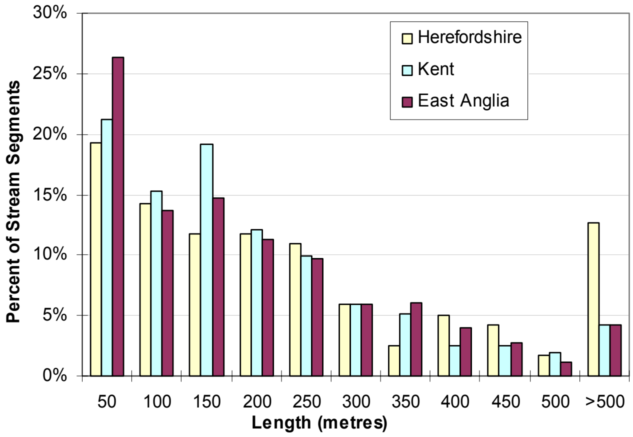


Figure 3.10: Distribution of stream segment lengths for Herefordshire, Kent and East Anglia

3.2.2 Presence of orchards

It is important to consider the presence of orchards near streams in conjunction with the hydrologic length and density discussed above. Table 3.7 shows OS MasterMap® orchard density to be highest in Kent, with 2,524 hectares of orchards within the five selected 5 x 5 km grid cells (125 km^2 total per area). East Anglia has the next highest orchard density at 1,525 hectares, and Herefordshire had 1,146 hectares of orchards. Although more OS MasterMap® orchards exist per unit area in Kent than in East Anglia or Herefordshire, a smaller proportion of the orchards have any portion of the field within 20 m of a stream in Kent than in East Anglia. Only five per cent of orchards in Herefordshire exist within 20 m of surface water, with 19 per cent in Kent and 32 per cent in East Anglia.

Table 3.7: Area of orchards within the grid cells for the three areas (either taken from OS MasterMap® or digitised from aerial images) and proportion of digitised orchards that have water within 20 m of some part of their area

Area	Area of all OS MasterMap orchards (ha)	Area of all digitised orchards (ha)	Area of digitised orchards with some part within 20 m of water (ha)	Area of digitised orchards as a % of area of OS Orchards	Area of digitised orchards with some part within 20 m of water as a % of area of digitised orchards	Area of digitised orchards with some part within 20 m of water as a % of area of OS Orchards
Herefordshire	1,146	743	38	65	5	3
Kent	2,524	1,018	196	40	19	8
East Anglia	1,525	942	302	62	32	20
Total	5,195	2,703	536	52	20	10

3.2.3 Stream perimeter and width of buffers

The high resolution imagery used in the classification process provided sufficient detail to classify the existence of even narrow buffers between orchard and water. However, the analysis indicated that there was a limited area of orchard (around two per cent) that was directly adjacent to water (with a buffer of less than one metre) in each of the three areas (Table 3.8). Closer examination of aerial imagery suggested that the few orchards where direct adjacency was identified mostly comprised large trees in small orchards. It is hypothesised that these orchards are either disused or under organic cultivation and thus not subject to the normal no-spray margin of three metres between the furthest edge of the treeline and the top of the bank.

Table 3.8: Selected percentiles of the distribution of direct adjacency

Site	Total stream segment count	Stream segments with any adjacency	Percent of total stream segments having adjacency
Herefordshire	119	2	1.7%
Kent	354	7	2.0%
East Anglia	1008	23	2.3%
All Water	1481	32	2.2%

Table 3.9 lists the number of measurements processed for each of the three sites. The number of stream segments is reported for each site, along with the total number of measurements originating at the perimeter sampling points, as well as how many of the total measurements were used to calculate the width of the buffer (that is, the shortest distance to crop for each sampling point). Note that the wind direction for each sample point was not always the same for the shortest distance to crop for buffer width calculations.

Table 3.9: Number of measurements to crop processed for Herefordshire, Kent and East Anglia

Site	Stream segment count	Number of measurements to crop	Number of shortest distance measurements to crop
Herefordshire	119	30,636	11,635
Kent	354	79,989	28,303
East Anglia	1,008	205,475	69,383
Total	1,481	316,100	109,321

Table 3.10 shows that the average width of the buffers, based on the combined measurements for the 119 Herefordshire stream segments, is 10.1 m using all directions (distances), and 8.9 m using only the shortest distances to crop. Average buffer widths for East Anglia are slightly smaller at 7.9 and 6.4 m, respectively. The average buffer widths for the Kent stream segments fall between those for the other two areas.

Table 3.10: Buffer width using all distances, and only shortest distance, to orchard

Area	Number of stream segments	Average buffer width using all measurements (m)	Average buffer width using only the shortest distance to orchard for each sampling point (m)
Herefordshire	119	10.1	8.9
Kent	354	8.8	7.4
East Anglia	1,008	7.9	6.4
All Water	1,481	8.3	6.9

Figure 3.11 shows the distributions of average buffer width for the water bodies in each of the three areas. The peak of the distribution for East Anglia occurs at 6-8 metre average buffer width and the histogram is concentrated more to the low end of buffer widths than those of Kent and Herefordshire. Peaks in the distributions occur at 8-10 and 10-12 m average buffer width for Kent and Herefordshire, respectively.

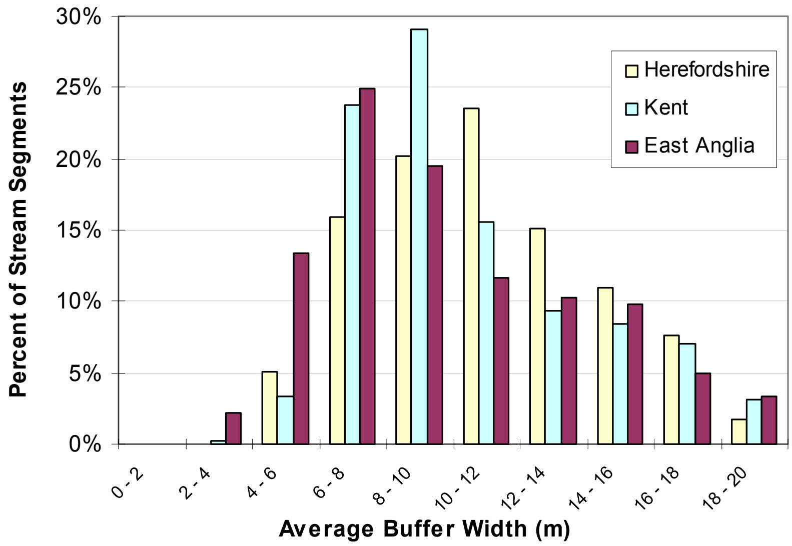


Figure 3.11: Average buffer width where orchards are within 20 m of streams

3.2.4 Buffer composition

Table 3.11 shows similar proportions of mitigating land cover in the buffer across the three areas. However, the individual classes of mitigating land cover are apportioned differently within the areas. Of the 119 stream segments in Herefordshire, 28% of the buffer is composed of forest trees, 9% hedgerow trees, and 2% scrub. The buffer in the Herefordshire sites is also composed of 52% grass, 7% urban/developed, 2% bare ground and small amounts of the remaining classes. Of the 1,008 stream segments in East Anglia, 7% of the buffer is composed of forest trees, 31% hedgerow trees, and 2% scrub. Note in this case, the forest class is used when a group of dense trees exist in a non-linear pattern.

Table 3.11: Buffer composition summarised by site (mitigating land cover is defined as forest + hedgerow + scrub; definitions of land cover types can be found in Section 2.3.2)

Site	Herefordshire	Kent	East Anglia	All Water
Number of stream segments	119	354	1,008	1,481
% Mitigating land cover	39	39	40	40
% Grass	52	56	53	54
% Trees: forest	28	23	7	14
% Trees: hedgerow	9	14	31	24
% Scrub	2	2	2	2
% Arable crop	<1	<1	<1	<1
% Other crop	<1	<1	<1	<1
% Urban/developed	7	4	4	4
% Bare ground	2	<1	<1	<1
% Other water	<1	<1	0	<1

3.2.5 Presence of mitigating land cover

Table 3.12 summarises information generated during the analysis of the presence of mitigating land cover. As described in Section 2.3.6, mitigating land cover was defined as trees (either forest or hedgerow) and scrub. Two ratios were calculated to compare the presence of mitigating land covers in the buffer across the three sites. The ratio of the count of transects intersecting at least one metre of mitigating land cover to the total count of transects indicates the proportion of the potentially exposed perimeter that will be protected by mitigating land cover (shown as 'percent mitigating transects' in Table 3.12). The ratio of the length of transect segments intersecting at least one metre of mitigating land cover to the total length of transects provides context to the first ratio, by determining the degree to which the mitigated perimeter is protected by mitigating land cover. Both ratios provide complementary descriptions of mitigation that exists within the three sites, and should therefore be considered jointly.

Table 3.12: Summary of transects counts passing through mitigating land cover

Site	Herefordshire	Kent	East Anglia	All Sites
Number of stream segments	119	354	1,008	1,481
Total number of measurements (transects)	30,636	79,989	205,475	316,100
Number of transects passing through mitigating land cover	18,020	41,603	98,886	158,509
Percent mitigating transects	59%	52%	48%	50%
Trees: forest (metres of transects)	86,796	160,341	109,148	356,285
Trees: hedgerow (metres of transects)	28,269	98,737	506,069	633,075
Scrub (metres of transects)	5,560	11,780	36,299	53,639
All mitigating vegetation (m of transects)	120,624	270,859	651,516	1,042,999
Average width of mitigating vegetation	5.8	5.6	4.8	--

Figure 3.12 and Figure 3.13 show the percentages of mitigating land cover compared to the entire buffer using the two methods described. In terms of the count of the transects intersecting mitigating land cover, Figure 3.12 shows that over 40 per cent of the stream segments in the Herefordshire site have 90-100 per cent of the potentially exposed perimeters protected by at least one metre of mitigating land cover. All sites demonstrate similar degrees of mitigation in terms of transect count, with the exception of the zero per cent mitigated class in which over 25 per cent of the stream segments in East Anglia do not have the presence of any mitigating land cover of at least one metre in width in the buffers.

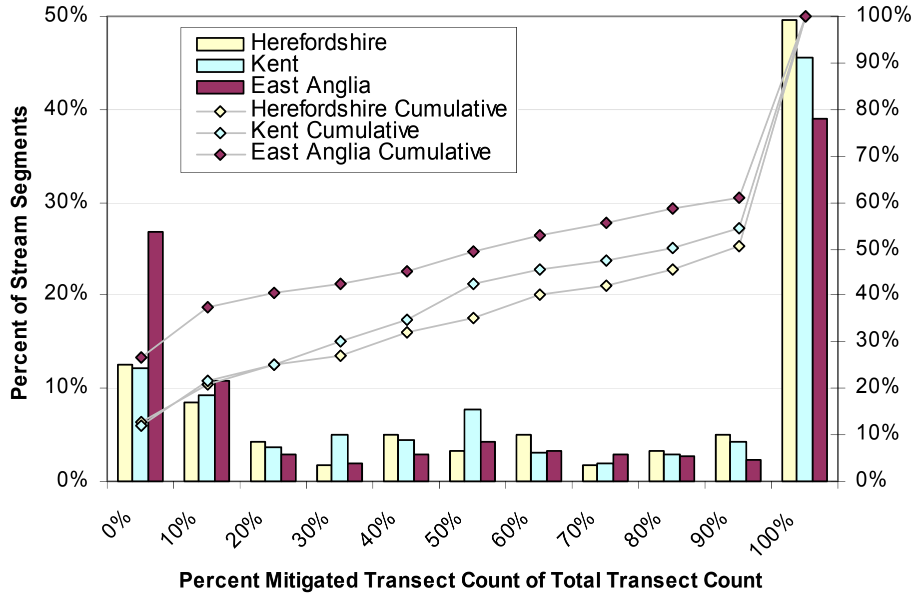


Figure 3.12: Buffer composition considering the count of mitigated transects to the count of total buffered transects (percent of transects that pass through at least one m of mitigating land cover; bars, left-hand axis; cumulative curves, right-hand axis)

Figure 3.13 examines the buffer composition in terms of the length of the buffered transects intersecting at least one metre of mitigating land cover to the total length of buffered transects. These results show a similar pattern to the number of transects passing through mitigating land cover. This reinforces the conclusion that East Anglia has a greater percentage of less protected water bodies than either Herefordshire or Kent. Over 25 per cent of the stream segments in East Anglia are not protected by any mitigating land cover in the buffer.

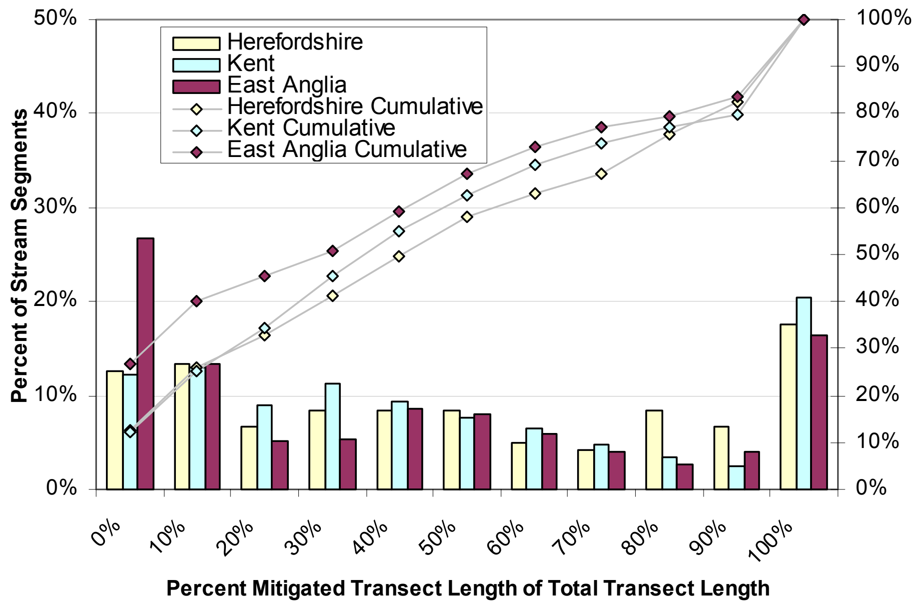


Figure 3.13: Buffer composition considering the length of mitigated segments of transects to the total length of all buffered transects (bars, left-hand axis; cumulative curves, right-hand axis)

Special attention was given to land covers that may help to mitigate spray drift. These land covers included forest, hedgerow and scrub in the land cover digitised from the aerial images. Considering <u>only</u> these mitigating land covers on each transect to calculate the average width of mitigating vegetation yields the data presented in Figure 3.14.

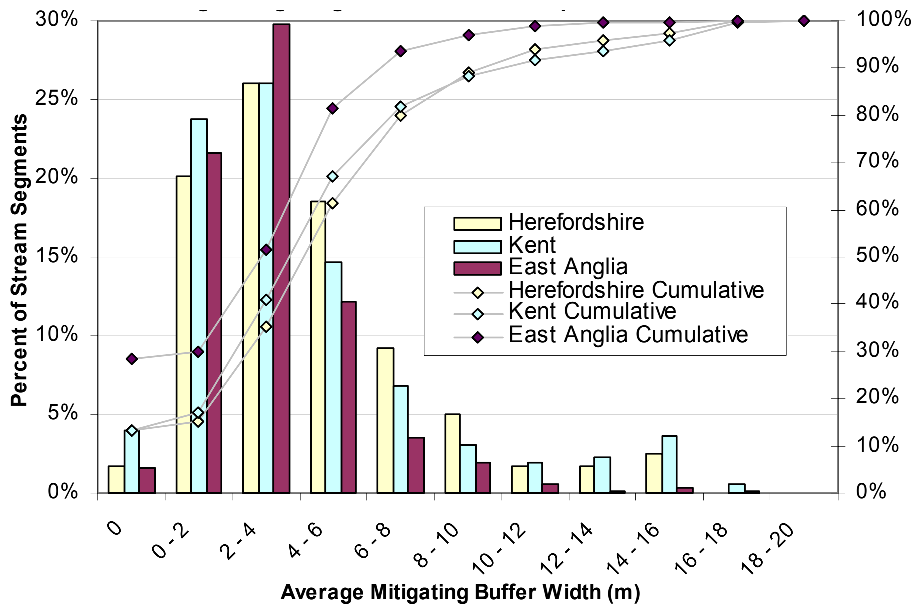


Figure 3.14: Average width of the mitigating vegetation present on buffers where crop is within 20 m of water (bars, left-hand axis; cumulative curves, right-hand axis)

3.2.6 Identification of the most exposed stream segments

The ratio of the count of transects intersecting at least one metre of mitigating land cover to the total count of transects indicates the proportion of the potentially exposed perimeter that will be protected by mitigating land cover (Figure 3.12). The stream segments having the greatest amount of potential exposure to orchard applications (having less than 10 per cent of the transects passing through one metre of mitigating land cover: the columns in the left-hand category – 0-10 per cent mitigated – of Figure 3.12) were mapped and are presented in Figures 3.15 to 3.17. Site visits (Section 3.5) included locations with low proportions of mitigation wherever possible.

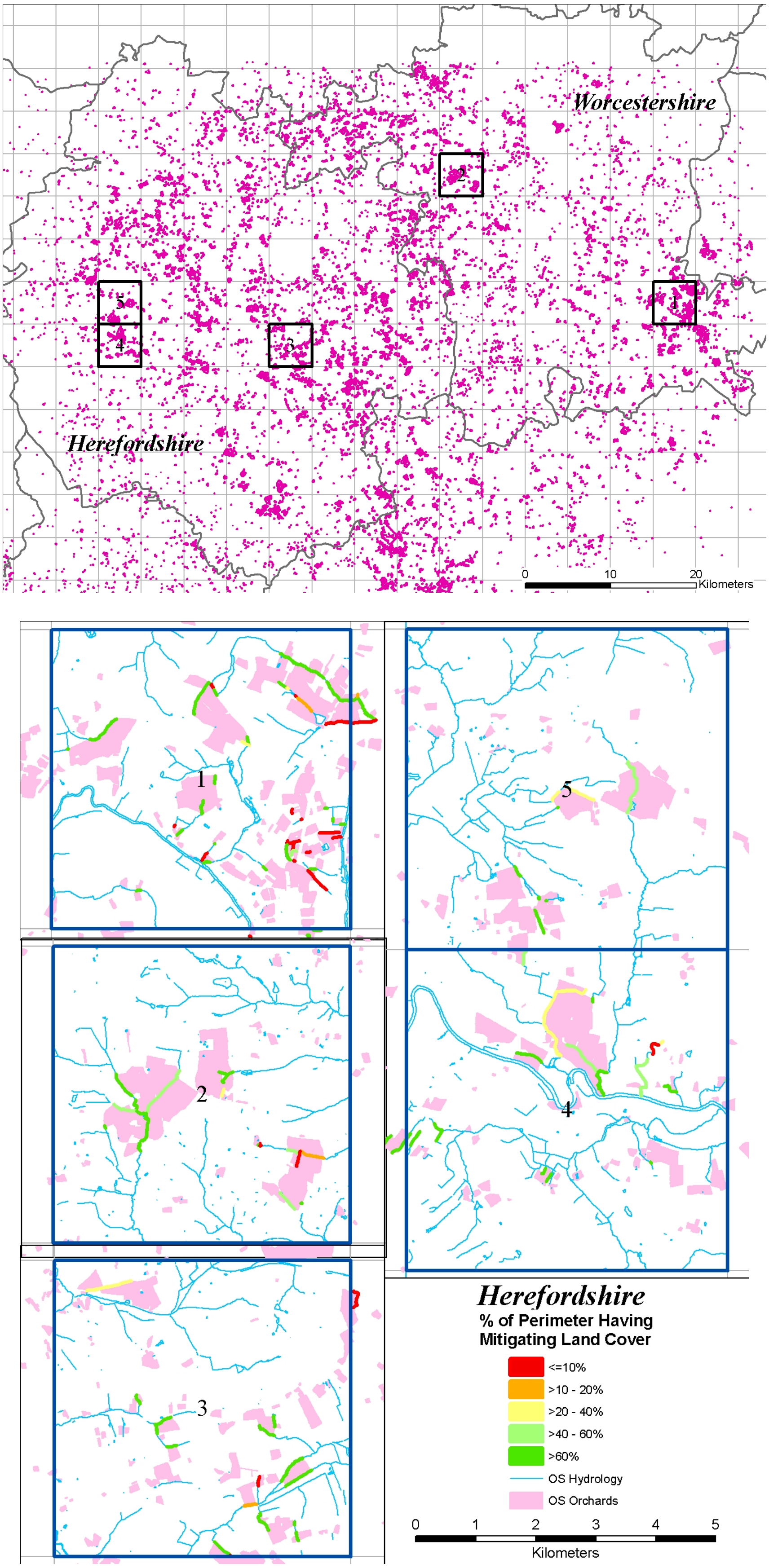


Figure 3.15: Map of Herefordshire showing the percent of stream perimeter having mitigating land cover

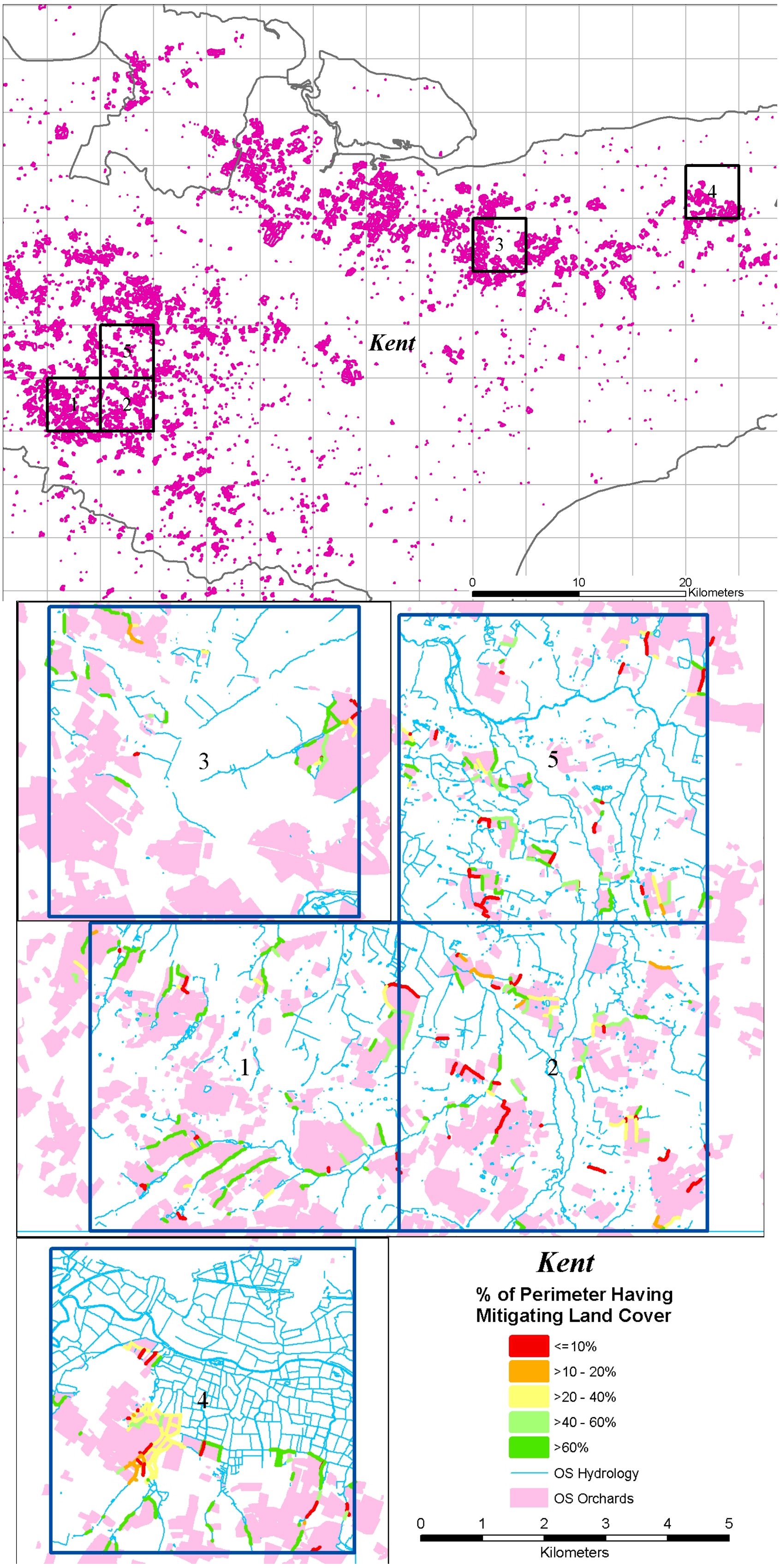


Figure 3.16: Map of Kent showing the percent of stream perimeter having mitigating land cover

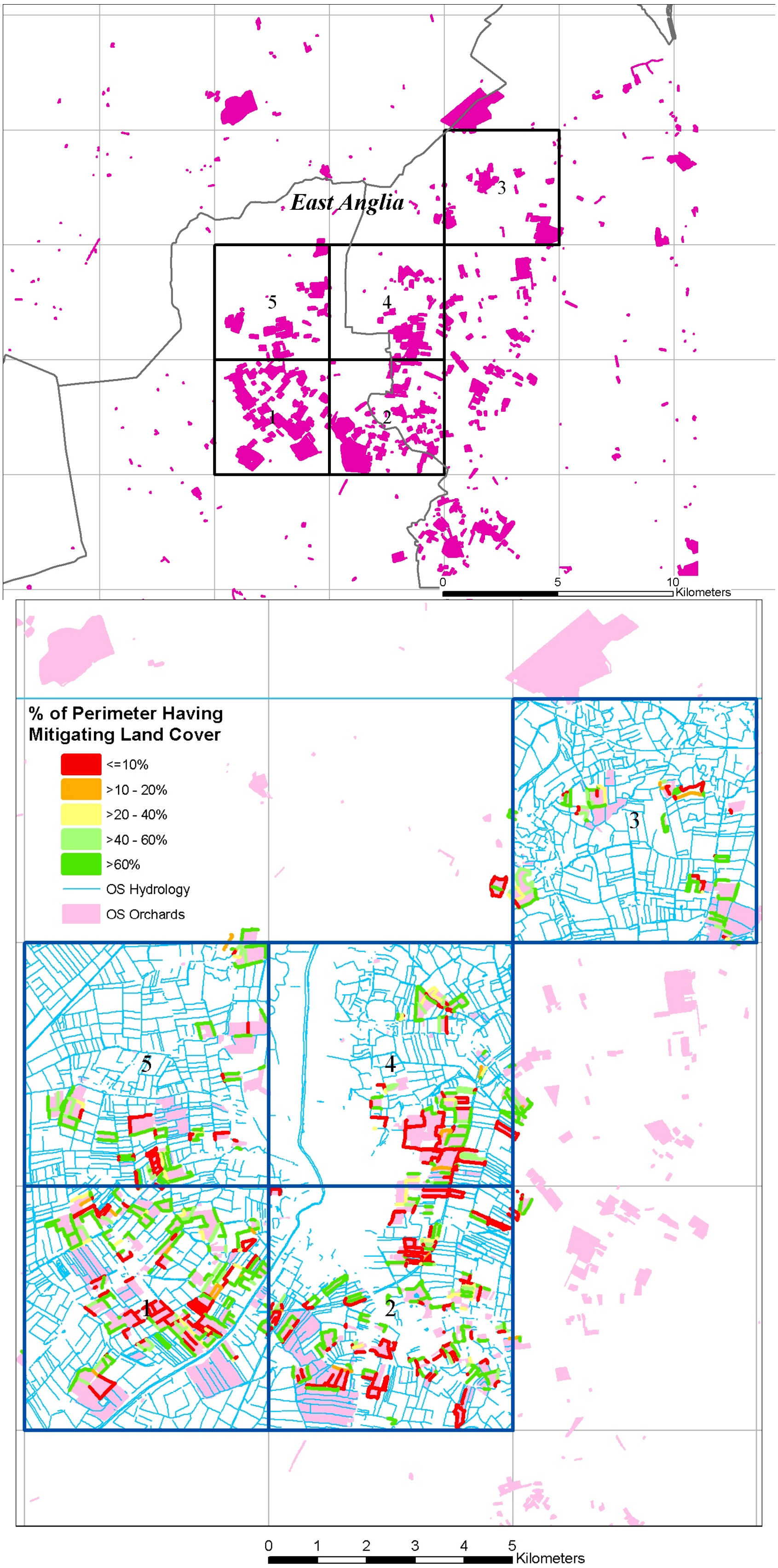


Figure 3.17: Map of East Anglia showing the percent of stream perimeter having mitigating land cover

3.3 Detailed analysis of the potential risk associated with pesticides used in orchards

3.3.1 Total toxic units per field

Risk was calculated for actual patterns of pesticide use on individual orchards in the Eastern region (selected because overall potential for exposure to pesticides was greatest in this region, Section 3.2). In contrast to national-level mapping, repeat applications with the same active ingredient were considered as separate events. Toxic units were calculated separately for *Daphnia* and algae for all individual treatments and the values for all chemical treatments were summed for each field. For example, if an orchard was treated with compounds A, B, C and D in March, April, June and September and the toxic units for *Daphnia* associated with the four applications was 0.2, 0.5, 0.1 and 1.0, then the summed risk to *Daphnia* on an annual basis was 1.8 toxic units. The summing did not consider the distribution in timing of individual applications. Summed potential risk for the 106 orchards is shown as a cumulative distribution in Figure 3.18. The summed risk to *Daphnia* was greater than that to algae in every case. Potential risk to *Daphnia* summed on an annual basis ranged from 0.02 to 72.6 toxic units, whilst that to algae ranged from 0.01 to 12.6 toxic units.

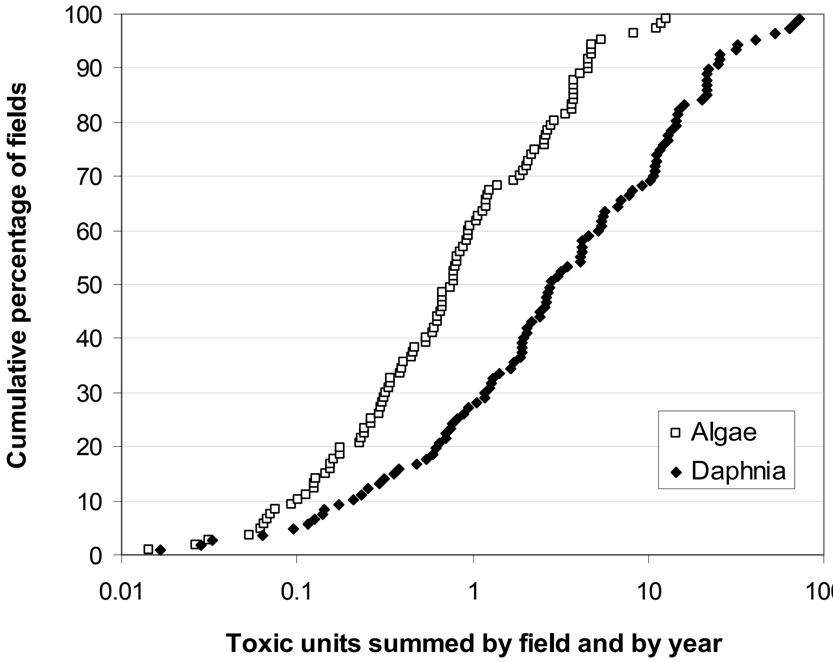
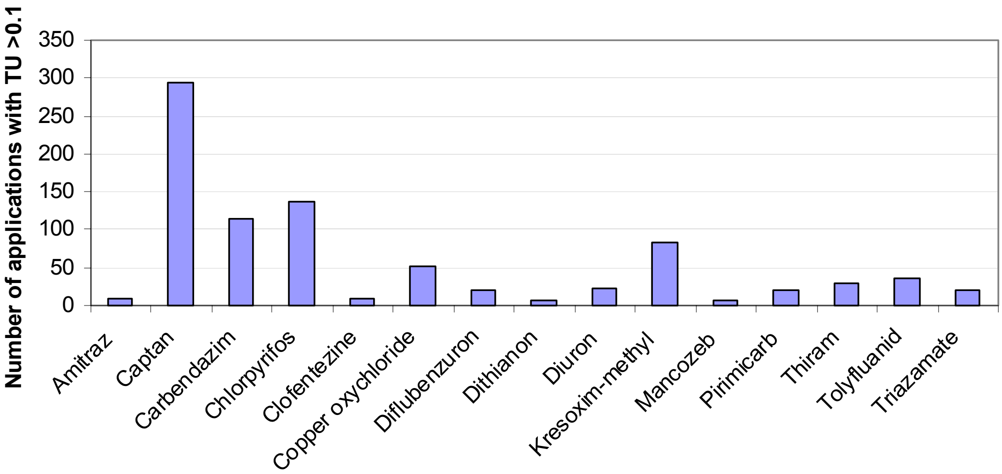


Figure 3.18: Cumulative distribution showing the potential risk for 106 orchards in Eastern region

3.3.2 Applications with the greatest predicted risk

A total of 861 of the 6,269 treatments with an individual active ingredient (14 per cent) gave predicted potential risk to *Daphnia* and/or algae of greater than 0.1 toxic units. These 861 treatments are shown divided by chemical and by time of application in Figure 3.19. Fifteen compounds are predicted to exceed 0.1 toxic unit on at least one occasion. The majority are insecticides and fungicides, but there is also an acaricide and a herbicide. Captan is the compound that is predicted to exceed 0.1 toxic units most frequently, followed by chlorpyrifos, carbendazim and kresoxim-methyl. The period during which applications predicted to have the greatest risk occur runs from mid-March to early August.



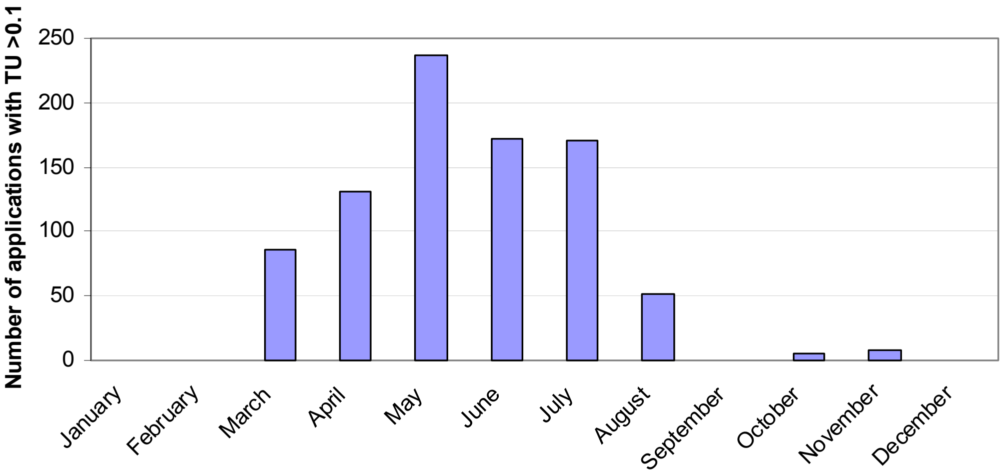
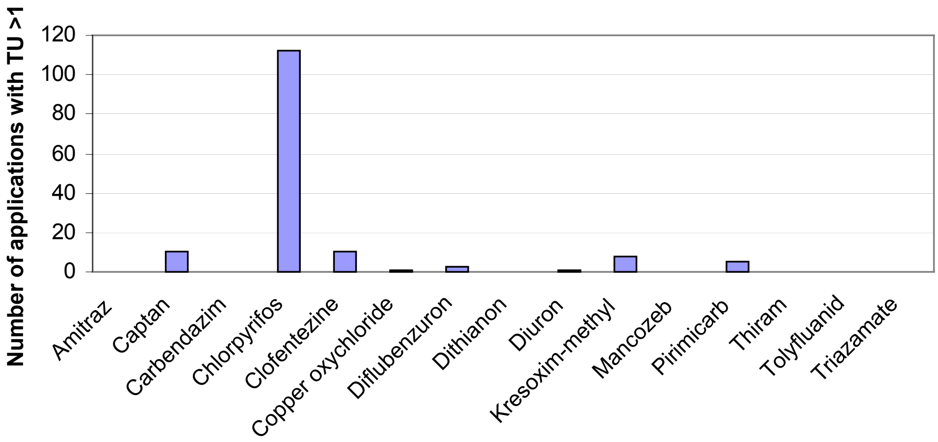


Figure 3.19: Chemicals (top chart) and time of application (bottom chart) for all treatments to 106 orchards in Eastern region predicted to give potential risk greater than 0.1 toxic units. Repeat applications with the same active ingredient are counted twice if the risk is predicted to exceed 0.1 toxic units in both instances

A total of 150 of the 6,269 treatments (2.5 per cent) gave predicted risk to *Daphnia* and/or algae of greater than one toxic unit. These 150 treatments are shown divided by chemical and by time of application in Figure 3.20. There are eight compounds that exceed one toxic unit on at least one occasion, again with the majority being insecticides and fungicides. Predicted exceedence of this threshold is dominated by chlorpyrifos. The period during which applications predicted to have the greatest risk occur runs from April to early August.



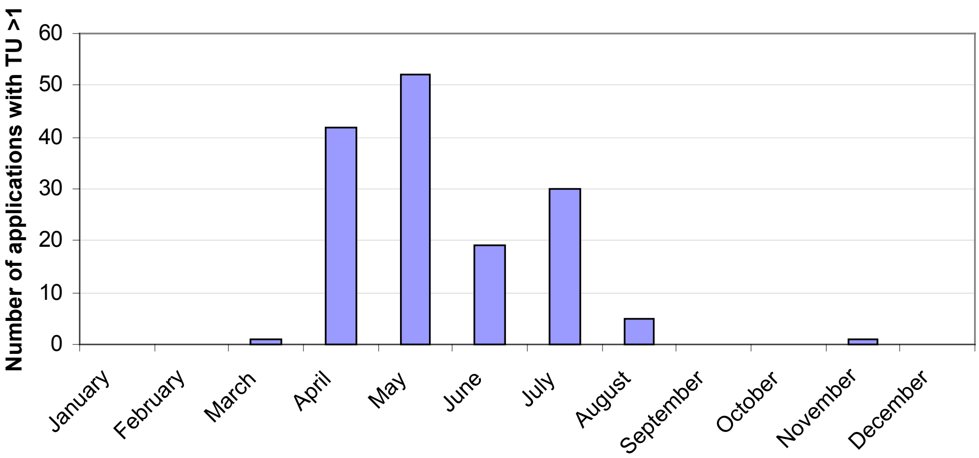


Figure 3.20: Chemicals (top chart) and time of application (bottom chart) for all treatments to 106 orchards in Eastern region predicted to give potential risk greater than one toxic unit. Repeat applications with the same active ingredient are counted twice if the risk is predicted to exceed one toxic unit in both instances

Dodine was identified as a compound causing significant potential risk in national-level mapping (Table 3.5). However, this compound does not appear in the 2004 usage dataset used for detailed analysis of risk associated with pesticide use on orchards. There are five products with dodine listed on the PSD database, all with 2008 expiry dates, so the switch from the use of dodine may be a crop management decision based on either product replacement or climatic conditions in 2004 (dodine is used to control scab).

3.4 Statistical design of sampling programme

The SOWAP dataset was selected for the main power analysis because it provides information on ecological assemblages at replicate sites for small streams close to the source. No other UK datasets were available with the same level of temporal replication for small streams with species-level taxonomic identification. This dataset shows significant variability between sites. For example, the mean species richness is eight with a coefficient of variation of 45 per cent and the mean species abundance is 319 with a coefficient of variation of 84 per cent . Variability is greater still when considering abundance of individual species. Results of the power analysis are summarised in Table 3.13 and details of the analysis are given for two examples in Figure 3.21.

Table 3.13: Summary of power analysis for the SOWAP dataset. The main body of the table shows the number of replicates per treatment required to detect the specified difference in taxonomic richness with statistical power of either 80 or 90 per cent

		Untransformed		Log transformed	
Test	**Power**	**All data**	**Species-level data only**	**All data**	**Species-level data only**
Based on t-test					
High *vs*. Low	80%	10	16	10	21
	90%	12	22	12	>25
Medium *vs*. Low	80%	25	>>25	>25	>>>25
	90%	>25	>>25	>>25	>>>25
Based on ANOVA					
40% difference	80%	10	20	-	-
	90%	12	25	-	-
25% difference	80%	>25	>>>25	-	-
	90%	>>25	>>>25	-	-

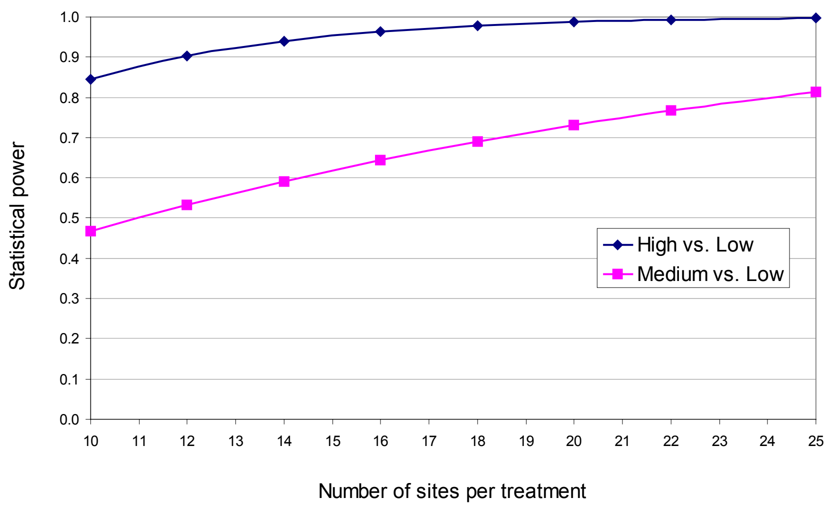


Figure 3.21: Power analysis based on t-test for the SOWAP dataset using data for all taxa and without transformation of the data (high *vs*. low and medium *vs*. low are for a 40 and 25 per cent reduction in taxa richness, respectively)

Based on univariate statistical techniques (t-test, ANOVA, linear regression), the power analysis indicates that it would be possible to have greater than 90 per cent probability of detecting a large (40 per cent) reduction in taxonomic richness within the constraints of a study based on 50 monitoring sites. For the specific dataset used for the analysis this would imply a reduction, for example from 20 taxa at sites with low exposure to pesticides to 12 taxa in sites with large exposure to pesticides. The probability of detecting the smaller impact (25 per cent reduction, 20 taxa reduced to 15 taxa) would be less than 80 per cent if only univariate techniques were used.

The power analysis was refined to investigate the possibility of including three treatments (levels of exposure) within the study. It was assumed that variance changed with the treatment, with the variance in highly impacted sites being 50 per cent of that in moderately and minimally impacted sites. Under these conditions, it would be possible to assign replicates with 20 reference sites, 20 moderately exposed sites and 10 highly exposed sites. The power to detect a 40 per cent difference in taxonomic richness between the highly exposed and reference sites would be 97.5 per cent, and the power to detect a 25 per cent difference in taxonomic richness between the moderately exposed and reference sites would be 81 per cent.

The dataset used in the power analysis is one of very few collected from water bodies directly relevant to the proposed study. Results of the analysis depend upon assumptions relating to the variance within different treatments. As there was no data available at the time of the analysis to support these assumptions on relative differences, the power analysis must be considered uncertain. Full analysis of the dataset collected in the proposed monitoring stage would draw on a range of statistical methods including multivariate techniques, Bayesian approaches and time series analysis. Each of these methods should increase the power to detect effects relative to the univariate power analysis reported above. However, more advanced power analysis (such as multivariate power analysis) is not possible given the lack of ecological data on which to base the work.

3.5 Site selection and site visits

Sites were visited in each of the three potential study locations, with qualitative assessments of suitability made at each potential site.

3.5.1 Site visits in Herefordshire

Herefordshire was the first area to be searched for potential field monitoring locations (Table 2.9). A total of 20 sites were visited (Figure 3.22). Visits were made to 14 priority sites determined from desk-based analysis of spatial data layers and aerial imagery. The desk analysis graded sites from one (very promising) to four (very unlikely to be suitable): six sites were graded one, nine were graded two, six were graded three, and 15 were graded four. A further six sites were visited that had been excluded by the GIS analysis because of presence of large buildings within 100 m of water, but which otherwise appeared to be promising.

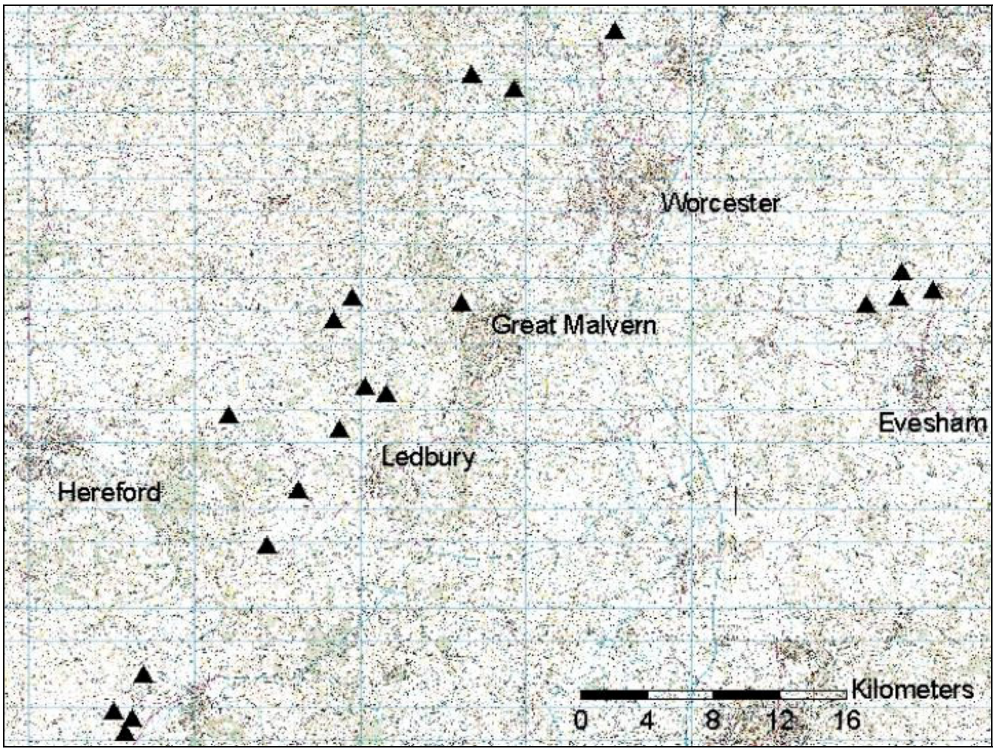


Figure 3.22: Map showing the location of potential monitoring sites visited in Herefordshire

Of the 20 sites visited, two were considered as potentially suitable locations for field monitoring. One was high priority based on the desk analysis (Site A; Plate 1). This site had potential for exposure via spray drift and there was some possibility that surface run-off might also occur. The second site (Site B; Plate 2) was excluded from the priority list because of adjacent buildings. There was evidence that surface run-off had occurred previously, but the water body was surrounded by trees and rough vegetation that would mitigate exposure via spray drift.

The remaining 18 of the 20 sites visited were unsuitable as locations for field monitoring. The reasons for exclusion were:

1. Some orchards had been grubbed up since the capture dates for aerial imagery and the OS MasterMap® orchard layer (2000-2002). There had also been a limited amount of new planting, although these orchards were at a very early stage in tree development.
2. Some of the water bodies selected as priorities in the desk analysis were ephemeral. This was a common reason for rejecting a site, because the specification for field monitoring required water to be present throughout the period when priority compounds were applied (primarily April to August, Figure 3.19). Comparison was made between 1:25,000 OS MasterMap® data on the GIS and 1:50,000 OS maps; generally streams shown on the 1:50,000 maps were permanent in those parts of Herefordshire visited. Streams shown on MasterMap but not on the 1:50,000 maps were too small.
3. Many of the sites were extremely well buffered from impacts from pesticides. Many had dense hedges and treelines that would preclude exposure via spray drift (Plate 3). Site management was normally bare soil beneath rows and thick grass between rows (Plate 4), so run-off along the rows of trees would be possible. However, all sites had a buffer of thick grass of at least five metres width between the last tree and the bankside vegetation, so any run-off would need to pass across this buffer before reaching surface water. One site had an adjacent field that was drained, but this was an arable field that was more low-lying than the orchard. It is unlikely that the orchard sites were drained, but

it would be necessary to check in winter when drains would be running and/or vegetation less dense to be absolutely sure.

4. Management of the orchards was variable. Some clearly received little management whereas others were very well maintained. Generally management was poorer in the smaller orchards and those to the south of the study area. It was assumed that pesticide application would follow management (being generally less intensive in orchards with poorer management).

Water samples were taken at four sites for basic chemical analysis. Table 3.14 shows that nitrate concentrations were acceptable at all four sites. Site B had low values for all measurements, whereas Site A had high values for phosphate and conductivity. Aquatic communities would probably be modified by high levels of background pollution, making it more difficult to observe effects from exposure to pesticides.

Table 3.14: Chemical analyses for water samples taken from sites in Herefordshire

Description	NO_3 (mg/l)	PO_4-P (mg/l)	Conductivity ($\mu S\ cm^{-1}$)	pH
Possible Site A	2.1 (Very low)	0.34 (Very high)	1730 (Extremely high)	8.1
Possible Site B	12.9 (Moderately low)	0.04 (Low)	730	7.8
Site highly buffered from pesticide impacts	10.6 (Moderately low)	0.00 (Low)	780	8.1
Site highly buffered from pesticide impacts	11.1 (Moderately low)	0.35 (Very high)	720	8.0

Assemblages of aquatic organisms were assessed briefly at nine of the 20 sites visited. The stream fauna was generally dominated by *Gammarus* sp. with a limited numbers of mayflies in the family Baetidae (such as *Baetis rhodani*) and caddis flies in the family Rhyacophilidae (*Rhyacophila* sp). Other components of the fauna were flatworms (*Polycelis* sp.), leeches (such as *Glossiphonia complanata*), water snails (*Radix baltica*, formerly known as *Lymnaea peregra*) and riffle beetles (*Elmis aenea*). All of the streams appeared to be affected by anthropogenic stresses (predominantly nutrients, but also other stresses that were not quantified). Several sites were either dry or were reduced to non-flowing pools.

Overall, the fauna included a moderate number of sensitive taxa indicative of streams that were mildly to severely stressed. The richness of taxa that are particularly sensitive to pesticides was low: there were few species of mayfly and caddis fly, and no stoneflies.

Thus, based on all measures of site suitability (potential for exposure, biology, water chemistry), only one of the 20 sites visited (Site B) was considered to meet the requirements for a monitoring location. As field visits focused on the most promising sites, it was concluded that visits to all streams adjacent to orchards in Herefordshire might double or treble the number of locations identified, but that the total number of possible sites for monitoring for high exposure to pesticides would probably not exceed five.

Follow-up desk analysis for Herefordshire

A revised GIS analysis was undertaken subsequent to the site visits in Herefordshire. This analysis started with the CEH water linework based on OS 1:50,000 mapping to ensure that all water bodies were permanent. The only selection criterion was that the stream should be within 50 m of an orchard that was at least five hectares in size. Using this decision process, a total of 109 streams were selected. Each of these sites was subsequently analysed individually by looking at OS data and aerial imagery. The refinement criteria considered size of water body (rivers were excluded), proximity to orchards and degree of buffering against spray drift. Anthropogenic impacts were not considered. On that basis, the analysis yielded:

- seven possible sites (including Sites A and B, though the first was rejected based on water chemistry);
- seven sites where at least one factor reduced their suitability as monitoring locations (such as scrubby vegetation on banksides), but where it would be appropriate to visit before discounting;
- 20 sites where one or more factors suggested that these locations would not be suitable;
- 65 sites that were definitely not suitable.

This second analysis was exhaustive in considering all water bodies of the appropriate size within the vicinity of orchards. The distance to orchards used to screen sites (50 m) was large and the analysis took no account of any grubbing of orchards in the last three to four years, of water chemistry at the sites or of potential impacts from roads, farms and dwellings upstream of the orchard. Based on the analyses, it was estimated that site selection for orchards in Herefordshire might yield about five potential sites where it would be worth talking to landowners about the possibility of monitoring.

Plate 1. Site A: New orchard planting on one side, older planting on the other; drift will be intercepted by thick bankside and riparian vegetation when present, but exposure will probably be greater in early spring

Plate 2. Site B: Stream is bordered by mix of thick vegetation, hedge and trees, but there is some evidence that surface run-off may be operating

Plate 3. Typical orchard site in Herefordshire – stream is completely buffered from drift by the thick hedge between sprayed area and water; area between rows and around edge of field is thick, permanent grass with no evidence of surface run-off moving to water

Plate 4. Typical cultivation system for orchards in Herefordshire – bare soil beneath trees, thick grass between rows, trees are 4 to 5 m high (probably around 3 m when pruned)

3.5.2 Site visits in East Anglia

There is intensive cultivation of orchards in an area surrounding Wisbech in East Anglia. This area was visited on two occasions to search for potential field monitoring sites. The first occasion focused entirely on likelihood of exposure of adjacent surface waters; possible sites were identified and water samples were taken for analysis of water quality. The most promising sites were then revisited to evaluate the biological communities present in ditches adjacent to orchards. Sites visited in East Anglia are shown in Figure 3.23.

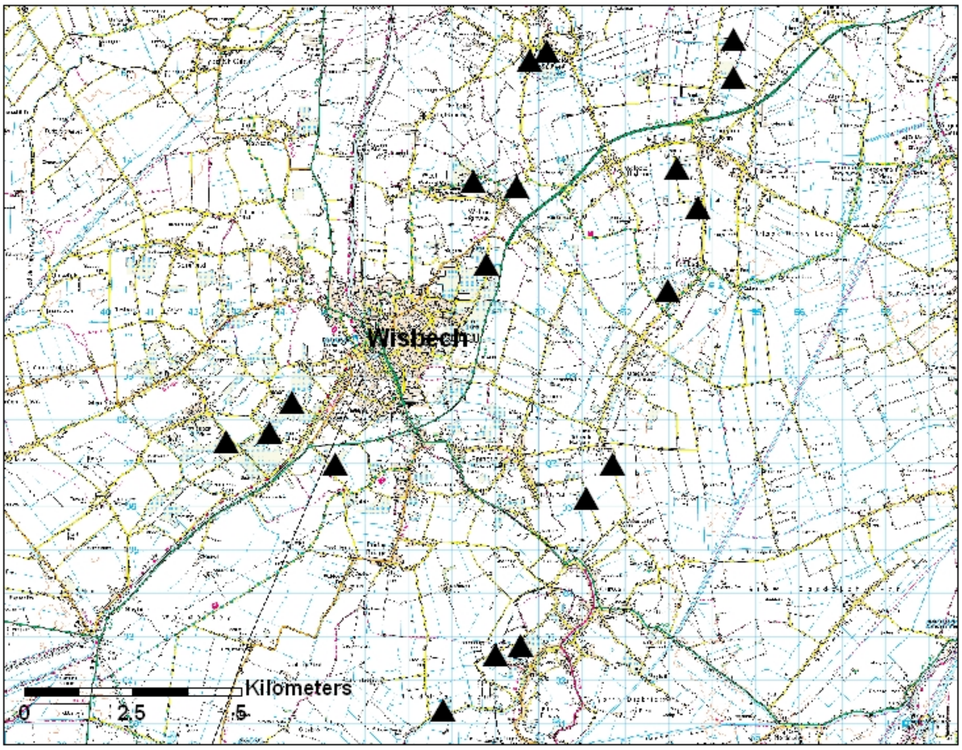


Figure 3.23: Map showing the location of potential monitoring sites visited in the vicinity of Wisbech

The initial visit in October 2005 focused solely on the suitability of sites based on potential for exposure. The fenland landscape is intensively drained, with drains of different sizes found at set depths within the landscape. The shallowest drains were about one metre below the soil surface and were found to be dry at the time of the visit. The next set were 1.5 to 2 m below the soil surface and contained water of 10-30 cm depth and around one metre width (Plate 5 – Plate 8). The third set of drains contained water of around 50 cm depth and 2-3 m width, whilst the final set consisted of the large, main drains leading to the River Nene. The second and third drain sets were considered as possible locations for the monitoring study. The dimensions of these ditches bracketed the dimensions assumed for calculating risk in this landscape (3 m wide, 30 cm deep; Table 2.3). A significant part of the water was covered with *Lemna spp*.

A total of 20 locations were visited in the vicinity of Wisbech and 11 were considered to be potential monitoring sites based solely on potential for exposure to pesticides. The orchards in this area were intensively managed. As in Herefordshire, the system was generally based on bare soil beneath tree rows with grass between and around the orchard (Plate 5). In contrast to Herefordshire, the banks of the ditches were maintained relatively free of vegetation, confirming the landscape analysis that showed Wisbech to have the largest potential for exposure to pesticides. Where present, vegetation between orchard and water was either well-maintained hedgerow (top of Plate 5) or windbreaks generally comprising poplars or other large trees (Plate 8). These windbreaks have relatively thin leaf cover near to ground level, so interception of spray drift may not be complete. In several instances, water bodies were found to run between an orchard and a field with either cereals or sugar beet.

Water samples were collected from five of the most promising sites for analysis of basic water quality measures (Table 3.15). All sites had extremely high conductivity, indicating a strong brackish influence. Concentrations of nitrate and phosphate varied considerably across the six sites, ranging from negligible to very high.

Table 3.15: Chemical analyses for water samples taken from sites in East Anglia

Description	NO_3 (mg/l)	PO_4-P (mg/l)	Conductivity ($\mu S\ cm^{-1}$)	pH
Location 1	[146.19]	[0.2046]	6,270 (Extremely high)	7.0
Location 2	6.202 (Low)	0.0726 (Moderate)	3,250 (Extremely high)	7.2
Location 3	0 (Very low)	0.5511 (Very high)	4,060 (Extremely high)	7.0
Location 4	[146.19]	[0.2046]	1,570 (Extremely high)	6.7
Location 5	6.645 (Low)	0.0627 (Moderate)	4,810 (Extremely high)	7.3

Samples from Locations 1 and 4 smelt strongly of hydrogen sulfide; analyses for nitrate and phosphate shown in square brackets are considered unreliable.

A visit to assess aquatic communities in the Wisbech ditches was carried out in January 2006. This visit focused particularly on the extent to which the brackish nature of ditches was associated with impoverished communities and whether organisms likely to be sensitive to pesticides were present. Although this visit took place outside of the most common periods for sampling invertebrate communities, previous experience suggested that sampling in spring would yield a larger number of organisms, but that broadly the same species would be expected to be present.

Aquatic communities were found to be significantly degraded, comprising: the salinity-tolerant amphipod *Gammarus zaddachi,* the isopod *Asellus aquaticus*, various gastropod molluscs (especially *Lymnaea peregra*, *Physa* spp. *Potamopyrgus antipodarum*, *Planorbis planorbis*, *Anisus vortex*, *Lymnaea palustris*), and various water beetles. Mayflies (*Cloeon* probably *dipterum*) were present in small numbers at only one site and a single damselfly in the family *Coenagrionidae* was found at one site. All sites were brackish, with conductivities ranging from 3,100 to 4,900 $\mu S\ cm^{-1}$ confirming the earlier observations made in October 2005. The brackish nature of the ditches was confirmed by the presence of the salinity-tolerant amphipod *Gammarus zaddachi* which was common in most locations. The brackish influence is probably related to the fact that the Wisbech area is drained by the River Nene, which is tidal approximately two kilometres upstream of Wisbech. Biological and landscape evidence (particularly complete cover of *Lemna* in some locations) also indicated that the ditches generally had high nutrient concentrations. This also confirms the results of earlier chemical analysis from October 2005.

Overall, the fauna consisted of invertebrates that are relatively insensitive to pesticides, with the exception of amphipods (although little is known of the specific sensitivity of *G. zaddachi*). Fauna was generally similar to that observed during a systematic programme of sampling around Spalding for Defra project PN0931. This earlier study also recorded assemblages that were generally pesticide tolerant in similar fenland ditches.

Subsequent to the field visit in January 2005, the orchard-growing area around Wisbech was discounted as a possible location for monitoring because the aquatic communities present were considered unrepresentative of freshwater systems across England and Wales. The following was concluded:

1. Wisbech ditches were brackish and so were not typical of the majority of freshwater environments of England and Wales.
2. Assemblages were dominated by pesticide-tolerant species
3. Ditches were generally heavily affected by nutrient pollution and consequently supported relatively impoverished assemblages, further exacerbating the general insensitivity of these assemblages to impacts from pesticide.

Plate 5. Orchard adjacent to ditch in the vicinity of Wisbech, East Anglia

Plate 6. Smaller ditch adjacent to orchard near Wisbech showing build-up of Lemna

Plate 7. Ditch adjacent to both orchard and cereals near Wisbech

Plate 8. Ditch near Wisbech with windbreak between orchard and ditch

3.5.3 Site visits in Kent

Kent was the last of the three possible study areas to be visited. Water quality samples were taken at the most promising sites but not analysed, as other criteria yielded enough information to reach a decision on suitability. Areas of orchard cultivation in Kent are relatively densely populated and there is a tight network of roads. Thus, anthropogenic influences were considered to be a significant limitation to possible monitoring locations. Sites visited in Kent are shown in Figure 3.24. Orchard cultivation in Kent occurs in two distinct landscapes. The area to the south west of Maidstone is an undulating landscape with till/loam soils. There is relatively little water in the vicinity of orchards in this area, particularly in parts with the greatest slopes. The other area of intense cultivation is on former marshes along the North Kent coast. This land is drained by a dense network of ditches and is very similar to the area around Wisbech in East Anglia. Visits were made to sites on both type of landscape.

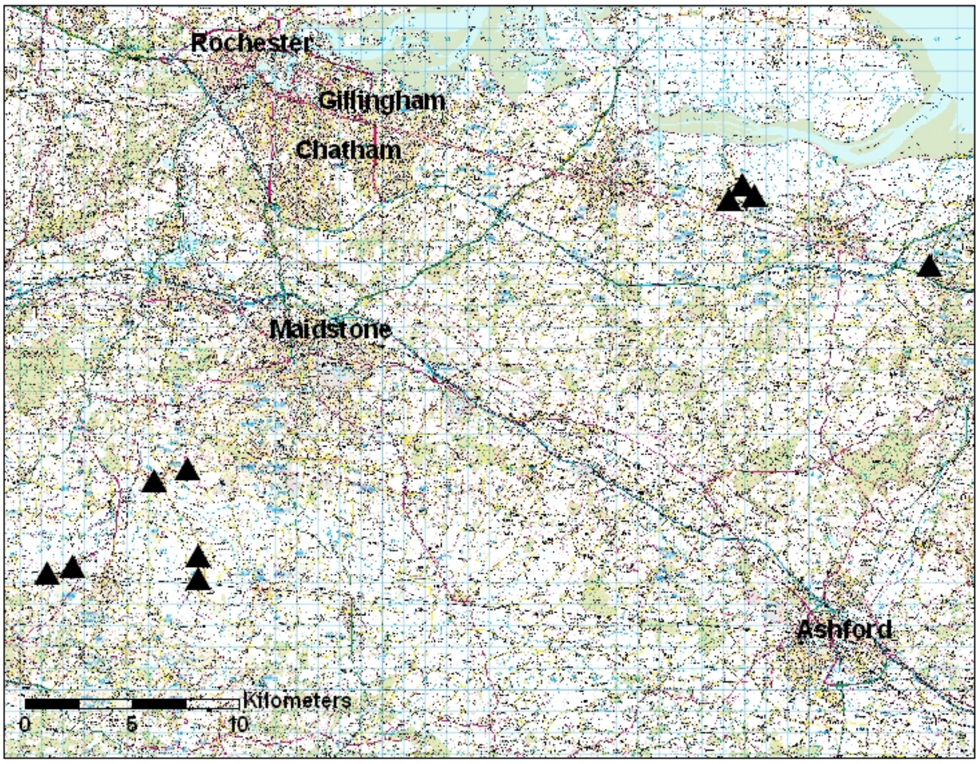


Figure 3.24: Map showing the location of potential monitoring sites visited in Kent

A total of 16 sites were visited in Kent. The sites to be visited were chosen by prioritisation and co-occurrence in relatively discrete areas. A significant amount of orchard had been grubbed up, particularly along the North Kent coast, and six sites were rejected for this reason. Four stream sites were found to the south west of Maidstone that had good potential for exposure to pesticides; in two cases the stream ran directly through the orchard (Plate 9, Plate 10), although there was a relatively large distance between trees and water. Three locations on the North Kent coast also had high potential for exposure to pesticides, but here the exposed water bodies were either large ditches (3-4 m wide, 0.7-1 m deep) or inland creeks (Plate 11, Plate 12).

Aquatic communities were assessed at eight of the sites visited, including examples of both streams and coastal ditches. The ditches had faunal assemblages dominated by amphipods (mainly *Gammarus* probably *pulex*), the isopod *Asellus aquaticus* and a range of fly larvae

(*Simuliidae*, *Chironomidae*, *Psychodidae*, *Dixidae*, *Tipulidae*, *Ceratopogonidae* and *Culicidae*). All of the sites supported at least one water snail family (normally *Lymnaeidae*, including *Radix balthica*) and pea mussels (*Pisidium spp*.), and most had several: *Hydrobiidae*, *Planorbidae*, *Bithyniidae*, *Physidae*, and *Valvatidae* were recorded. Most of the sites supported leeches of various sorts (families *Erpobdellidae*, *Glossiphoniidae*, and *Piscicolidae*).

All of the sites supported cased caddis flies in the family *Limnephilidae*; caseless caddises in the family *Hydropsychidae* were found at a single site. Mayflies in the family *Baetidae* (probably *Baetis rhodani*) were found in three sites but only as single individuals. Damselflies (family *Coenagriidae*) were found in one site.

Overall, the ditches supported assemblages dominated by species less sensitive to pesticides than *Daphnia magna*, although taxa may have greater sensitivity to specific compounds. The streams had some pesticide sensitive taxa, although only a small number of species and in low abundance: low numbers of mayflies and caddis flies, and no stoneflies. None of the pesticide sensitive taxa were a dominant component of the fauna, probably because of a combination of water quality stress and the small size of the streams.

In discussion with the sponsors, potential monitoring locations identified along the North Kent coast were rejected because the aquatic communities present were impoverished and generally insensitive to pesticides. Locations to the south west of Maidstone were more promising and two of the four sites with reasonable exposure also had moderate biology. Further GIS analysis after the field visits identified a maximum of eight further stream locations that had not been visited. Based on the rejection rate during site visits in Kent, it was estimated that site selection for orchards in Kent might yield about five potential sites where it would be worth talking to landowners about the possibility of monitoring.

Plate 9. Stream to the south west of Maidstone in Kent. The stream runs between two orchards. There is little bankside vegetation to intercept spray drift, but distances from the trees to the stream are relatively large

Plate 10. Stream to the south west of Maidstone in Kent. This stream runs through an orchard, but distance from stream to water is relatively large. The orchard is fully grassed beneath trees

Plate 11. Inland creek adjacent to an orchard on the edge of the North Kent marshes

Plate 12. Additional view of inland creek adjacent to an orchard on the edge of the North Kent marshes

4 Analysis and discussion

4.1 Relative risks to the aquatic environment from agricultural use of pesticides

National-level risk mapping clearly identified that crop is the major driver for potential risk to the aquatic environment from the use of agricultural pesticides. There were two orders of magnitude difference in potential risk between the highest and lowest ranked crops. In contrast, differences in risk for the same crop grown on different landscape classes or regions were smaller, though still significant. Top fruit and hops were identified as the two crops associated with the greatest risk. Soft fruit and bulbs had intermediate risk, but predicted risk for all other crops was an order of magnitude smaller than for top fruit and hops.

The risk calculation was based on a toxic unit approach (where risk was expressed as the proportion of an LC/EC50 value predicted to be present in surface waters). The calculations aggregated the maximum exposure value for all pesticides, combining multiple applications of a single active ingredient, normalised to the toxicity for that pesticide, and then simply summed the values for all compounds applied to a particular crop independent of timing of application. Worst-case assumptions were taken from regulatory risk assessment (such as minimum buffers between crop and water), although some assumptions were refined to reflect knowledge about a particular landscape (such as dimensions of streams in different landscapes). Mandatory no-spray buffers were included in the calculation of spray drift where applicable. Despite the simplifications in the approach, it is notable that aggregated risk did not exceed one toxic unit for the majority of arable crops in the majority of situations. Figure 3.1 shows that this threshold of one toxic unit was exceeded in every situation for top fruit and hops. For *Daphnia magna*, one toxic unit was sometimes exceeded for bulbs, hardy nursery stock, oilseeds, other stockfeeding, other vegetables, dry peas and soft fruit, but the median was only above this value for soft fruit. For algae, one toxic unit was sometimes exceeded for bulbs, cereals, hardy nursery stock, oilseeds, other vegetables, potatoes and soft fruit, with the median value close to one for other vegetables and below one for the remainder. These calculations are per area of crop.

Maps of risk (Figures 3.2 to 3.4) clearly reflected the importance of orchards in risk to the aquatic environment. Most intense areas of potential risk were located in Herefordshire and Kent. This clustering of risk was more intense for calculations based on toxicity to *Daphnia magna* than for algae. Potential risk was more evenly distributed across the country for algae, primarily because of the more widespread application of high usage herbicides on crops such as cereals and oilseeds. The maps were derived by averaging risk across the 6,600 catchments identified by the Environment Agency. On this basis, less than four per cent of catchments had average risk to either indicator organism of greater than one toxic unit.

There are two versions of each risk map. The first considered pesticide usage data collected between 2000 and 2003. Significant risks were found to be associated with compounds that had been withdrawn in the intervening period (either withdrawn or in a phase-out period). These compounds were identified and a second set of maps was constructed excluding the risk from these compounds. It was not possible to identify the usage of new or existing compounds that had replaced the withdrawn ones. It is thus likely that the first set of maps overestimates the potential risk from current usage and the second set underestimates the risk. A more accurate reflection of changes in predicted risk over time could be obtained by repeating the mapping exercise as new pesticide usage data becomes available. Despite these shortcomings, the two sets of maps suggest the positive influence of the review process in reducing risk. Differences are particularly marked for the higher risk crops and the reduction was predicted to be an order of magnitude for risks to *Daphnia magna* associated with hops, due to the withdrawal of tar oil previously used as a defoliant.

Risk for each crop-region-landscape combination was predicted to be dominated by a small number of pesticides (Table 3.5). A field-level analysis of risk associated with the use of pesticides on orchards in the Eastern region showed that 2.5 per cent of all applications were predicted to have potential risk of greater than one toxic unit. These applications included three insecticides, two fungicides, an acaricide and a herbicide, but the dataset was dominated by a single insecticide (chlorpyrifos). Applications giving potential risk greater than one toxic unit were most frequent in April and May and occurred predominantly between April and July.

The national-level mapping is useful to focus attention on locations and situations associated with the greatest potential risk to the aquatic environment. Nevertheless, these calculations only show potential risk and assumptions need to be considered when interpreting the output:

- Risk is averaged across the catchments that are the base unit for mapping in Figures 3.2 to 3.4. For example, a catchment comprising 50 per cent cereals with risk of 0.5 toxic units and 50 per cent non-agricultural land with no risk would have average risk of 0.25 toxic units.
- It is assumed that all fields have an adjacent stream with dimensions equal to the average for the respective landscape. Given that risk is mapped on a dimensionless scale, the assumption is not limiting provided that the streams present are equally distributed between the weighted area of different crops and non-cropped land. No account is taken of the connectivity of streams and transfer of pesticide downstream.
- All applications of a single active ingredient are lumped into a single application. Maximum exposure and thus maximum risk is calculated for each active and these are summed on a per-crop basis independent of timing of application.
- Exposure calculations are based on regulatory assumptions that will not always hold in practice. For example, the distance from arable fields and orchards to water is assumed to be one and three metres respectively; spray drift is calculated as a 90^{th} percentile value to fit in with regulatory practice. No account is taken of actual distances from crop to water, or of intervening vegetation that may present a barrier to spray drift
- It is assumed that *Daphnia magna* and algae (normally *Scenedesmus*) are good indicators for toxicity to aquatic communities of invertebrates and plants. Further, it is assumed that combining the maximum predicted exposure concentration with the output from laboratory test protocols provides an adequate expression of risk in field situations. Further work on the exposure and/or ecotoxicity of a pesticide would be triggered by an identification of risk based on first-tier calculations. All pesticides within the second set of maps (those excluding withdrawn compounds) are registered for use and thus have data demonstrating that risks are acceptable according to the scheme in operation at the latest time of review. No account is taken of this additional, higher-tier information.

4.2 Local conditions associated with cultivation of orchards

National-level mapping identified orchards as the crop type associated with the greatest potential risk. Landscape-level analysis was then undertaken on individual stream segments to investigate local characteristics of orchard cultivation and thus to determine how these might influence exposure to pesticides. Three regions were identified for this analysis and five 5 x 5 km grid cells were selected in each area as having the densest cultivation of orchards near water. All stream segments that were located within 20 m of some part of an orchard were selected for landscape analysis (1,481 segments in total). Throughout all analyses, the areas in Herefordshire and East Anglia were at opposite ends of the distribution of results, with Kent intermediate. This was because orchards in Kent are grown in two contrasting landscapes that are similar to those found in Herefordshire and East Anglia respectively.

National risk mapping was based on the assumption of a three metre buffer distance between the edge of the treeline and water. In reality, there were almost no buffers that were smaller than four metres in the areas investigated (Figure 3.11). The modes of the distribution of buffer widths were 6-8, 8-10 and 10-12 m for East Anglia, Kent and Herefordshire, respectively. Buffer characterisations showed a small amount of orchard (around two per cent) that was directly adjacent to water in each of the three areas, but the aerial images suggest that this was probably attributable to disused orchards.

Calculation of exposure via spray drift for national mapping assumed that streams were completely unprotected by any bankside vegetation. Landscape-level analysis took a conservative approach to the definition of bankside vegetation that will intercept a portion of the spray drift, including only trees, hedges and dense scrub. The extent to which individual stream segments were protected by one of these three forms of mitigating vegetation was bimodal in distribution (Figure 3.12). Nearly half of all stream segments located within 20 m of orchards had protection from mitigating vegetation along the full length of the segment and considering all possible directions from crop to water. Nevertheless, a significant number of stream segments (more than 10 per cent in Herefordshire and Kent, greater than 25 per cent in East Anglia) did not have any of the mitigating vegetation types between crop and water. The distribution shifted towards lower levels of mitigation when expressed on a per unit length basis rather than per individual stream segment (Figure 3.13).

Taken together, the analyses of buffer widths and bankside vegetation give a perspective on how exposure to pesticides is likely to vary relative to that predicted in national-level mapping where worst case assumptions are made. Mapping of the stream lengths with the least mitigation (Figure 3.15 to 3.17) clearly shows the power of the GIS analysis in targeting locations to be considered in any monitoring programme. The sites in East Anglia are highlighted as having a far greater density of stream segments adjacent to orchards, with generally lower levels of mitigation by bankside vegetation. This identification of high exposure sites could be strengthened by using a drift model to incorporate the influences of both buffer width and interception by bankside vegetation, and by considering the likelihood for drift passing over adjacent waters based on probabilities of occurrence of specific wind directions.

4.3 Statistical design of the monitoring study

Statistical analysis to determine the optimum design for the monitoring study was limited by two constraints, the first of which was the availability of a dataset analogous to that to be generated during the study. Power analyses would ideally have been based on a dataset that characterised variability in space and time in macroinvertebrate and diatom communities in small streams in the presence and absence of an impact. In the absence of such a dataset, assumptions had to be made about the level of impact that could be expected from exposure to pesticides and about how variability would compare in affected and non-affected locations. A second constraint was that the power analysis was based solely on a number of univariate techniques. Statistical analysis of the dataset generated during the monitoring phase would involve univariate and multivariate methods, as well as making use of Bayesian approaches and time series analysis. Multivariate approaches would increase the level of statistical power due to correlations between taxa present at replicate monitoring locations, but it was not possible to determine the extent of this increase in power without a full dataset on which to base the calculations.

The project initially specified a total of 50 sites to be monitored in the field phase, so the analyses focused on determining the power to detect effects based on this sample size. Calculations investigated whether the sites should be split into two or more classes of exposure (high/low, high/medium/low and so on) and whether it would be possible to split the sites further between two different regions. Statistical power was set to a preferred value of 90 per cent and a minimum value of 80 per cent; 90 per cent power implies that if ten studies were run in parallel, then on average nine of those studies would detect the effect being targeted. The optimal study design assigned replicates with 20 reference sites, 20 moderately exposed sites and 10 highly exposed sites. The power to detect a 40 per cent difference in taxonomic richness between the highly exposed and reference sites was calculated to be 97.5 per cent, and the power to detect a 25 per cent difference in taxonomic richness between the moderately exposed and reference sites would be 81 per cent.

Taxonomic (species) richness is an aggregated measure of biodiversity that is rather crude, but that reduces the variability between replicate sites. The impacts that could be detected using only univariate methods are large, assuming that variability in the SOWAP dataset is similar to that at the study locations. An important reason for this is the monitoring-type study design whereby sites are allocated to exposure classes and aquatic assemblages are compared across classes. The alternative would be to adopt a paired comparison structure, as was used in the GMHT Farm Scale Evaluation Trials (Perry *et al.*, 2003). Pairing might be achieved by matching high impact sites with similar locations adjacent to non-agricultural land or buffered by natural vegetation or by a fence erected for the purposes of the study to prevent transfer of spray drift. A simpler approach would be to locate control sites upstream of the target site (such as upstream from orchards). The principle advantage of the paired approach is that statistical comparisons are made between paired locations rather than across replicate sites within treatments, so some of the variability in the latter analysis is avoided.

4.4 Availability of monitoring locations with high exposure to pesticides

Site selection focused on the highest potential risk situation identified in national-level risk mapping (orchard cultivation). Table 4.1 summarises the estimated availability of sites for field monitoring that would fall into the highest risk category.

Table 4.1: Summary of availability of monitoring locations with high exposure to pesticides and meeting other specifications for site selection

Area	Estimated number of high-exposure sites	Exposure	Biology	Water chemistry
Herefordshire	5	+	++	++
East Anglia (Wisbech)	15	+++	-	-
Kent (near Maidstone)	5	++	+	?
Kent (North Kent coast)	5	+++	-	?

+ symbols indicate the extent to which the sites met the respective criterion (+++ is the best match)

- symbols indicate that the site were rejected on the basis of the criterion

? symbols indicate absence of information for the respective criterion

No sites were found that completely matched all of the specification for study locations. Ditch sites near Wisbech and along the North Kent coast were rejected because the aquatic communities lacked taxa sensitive to pesticides and were significantly affected by agricultural pollutants. The sites identified in Herefordshire and near Maidstone offered the best compromise between exposure conditions, presence of a representative community including some pesticide sensitive taxa and water chemistry indicating a moderate level of background pollution from agriculture and other anthropogenic inputs. It was estimated that around five sites matching all of the criteria could be located in each of these areas. No approach to landowners was made during the site visits, and some sites were expected to be withheld from use by the landowner because of the relatively sensitive nature of the monitoring to be undertaken.

5 Conclusions

5.1 Decision on whether to proceed to the field monitoring phase of the project

The results presented in Section 4 were used as the basis for deciding whether or not to proceed to field monitoring of exposure to pesticides and any associated effects on aquatic invertebrates and plants. In the light of these results, the project consortium and project steering committee decided that it would not be appropriate to continue to the monitoring-based stage. The reasons for this were as follows:

1. Risk mapping clearly identified orchards as the crop type associated with the greatest potential risk to the aquatic environment. However, landscape analysis using GIS datasets and aerial imagery coupled with site visits showed that conditions associated with the greatest exposure occurred very infrequently. In locations where high exposure was anticipated, aquatic ecosystems generally lacked pesticide sensitive taxa as a result of stresses related to aspect, water body type or agricultural pollution, such that sensitivity to pesticides was considered low relative to other parts of England and Wales.
2. It was decided that it would not be possible to obtain a sufficiently replicated sample of sites representing high exposure to pesticides coupled with a sensitive aquatic community. Alternative crops with greatest potential risk (hops, soft fruit, bulbs) have very small areas of cultivation. Widespread crops that rank next on the list (vegetables, potatoes, oilseeds) were predicted to have more than an order of magnitude smaller risk than orchards. Thus, monitoring of water adjacent to these lower risk crops would not meet a central objective of the study to investigate the highest risk situations.
3. There is great natural variation in aquatic communities in space and time. Statistical analyses indicated that the monitoring design would only have sufficient power to detect relatively large changes in aggregated measures of biodiversity. For example, analysis based on the best UK dataset yielded a design with 81% power to detect a 25% reduction in invertebrate taxonomic richness and 98% power to detect a 40% reduction in taxon richness. This design assigned 10 monitoring sites with high exposure to pesticides and 20 each with moderate and low exposure. Use of multivariate techniques would strengthen statistical power, but the extent of this improvement was unknown.
4. The project originally specified the use of passive sampling devices to provide integrative measures of exposure to pesticides over time. The Environment Agency tested these devices in parallel with the work reported here (Environment Agency, 2006). Whilst a large number of pesticides could be monitored using the devices available, there were significant gaps in the suite of pesticides that could be picked up and these gaps were not readily explained based on compound chemistry. Primary exposure arising from the use of pesticides on orchards was expected to be via spray drift and residence time in stream systems was expected to be very short (possibly five to 15 minutes). Tests based on short-term exposure to concentrations equivalent to worst-case predicted environmental concentrations showed that there would be problems in detecting some of the priority pesticides for this exposure pattern.

The decision was thus taken to terminate the project at the end of the design phase (Task 1). Nevertheless, the results presented in this report go some way to addressing the question of whether the use of agricultural pesticides under field conditions is associated with adverse effects on aquatic invertebrates and plants. Crops associated with the greatest potential risk are all relatively specialised and have small areas of cultivation. Orchard cultivation is by far the most widespread of these high-risk crops. However, locations and conditions of cultivation are such that levels of exposure to pesticides are very much lower than predictions using standard assumptions from risk assessment. Furthermore, water bodies where greatest exposure to pesticides is anticipated are lowland ditch systems, where the aquatic community is often impoverished and thus relatively insensitive to effects from exposure to pesticides.

Despite the progress made here, the central question of this research remains essentially unanswered. For example, it is not possible to state whether, and to what extent, exposure to pesticides has led to the degradation of aquatic communities. Determining whether the use of pesticides is associated with adverse effects on aquatic invertebrates and plants has important implications for the way in which pesticide use is regulated and monitored in the aquatic environment. It is thus important to consider alternative approaches to meeting the overall aim of this research, rather than to abandon the attempt to address the question.

6 Recommendations

The recommendations made here are those of the research consortium. They will be considered by the Environment Agency and the Pesticides Safety Directorate.

6.1 Use of risk mapping information

National-level risk mapping has been used here to identify specific locations, crops and active ingredients associated with the greatest potential risk to aquatic invertebrates and plants. This approach can be used to underpin a risk-based monitoring strategy for the aquatic environment. Maximum value will be obtained by interrogating the datasets produced on specific objectives and/or questions. Consideration should be given to extending the exposure component of the mapping to include transfer of pesticides to the aquatic environment via surface run-off and to include Wales as cropping data becomes available. The exercise undertaken here provides a baseline for aggregated risk; the exercise could be repeated as new information is generated, to demonstrate changes in risk resulting from policy initiatives (such as a review of existing plant protection products) and from changes in cropping, pest pressures and pesticide use.

Landscape-level analysis was used to characterise the specific conditions associated with cultivation of orchards. This analysis can be a powerful alternative line of evidence when considering the potential effects of pesticides in the field. Consideration should be given to extending the approach to investigate a range of crop types other than orchards. This would reinforce existing regulatory approaches by demonstrating how frequently worst case assumptions built into the risk assessment actually occur in practice. It would also help to refine any risk-based monitoring strategy.

6.2 Future investigations into effects of pesticides in the field

The aim of this project was to establish whether the use of agricultural pesticides adversely affects aquatic invertebrates and plants. The consortium behind this project considers that addressing this aim remains extremely important from both scientific and policy perspectives. The recommendation of the consortium is that the aim can only be addressed by bringing together worst case exposure to pesticides and a sensitive assemblage. Field visits showed that the two components do not co-occur to a significant extent, so this combination would need to be created experimentally. The study design outlined below could help to address the central aim of the project:

1. Monitor pesticide concentrations in a high exposure situation (such as the Wisbech ditches adjacent to orchards) for a season. Risk mapping demonstrates that this would give a reasonable representation of worst case exposure for UK conditions. Active and passive sampling systems provide complementary data, so an ideal approach would combine automatic water samplers to characterise magnitude and duration of peaks in exposure and passive samplers to measure total exposure. To capture short-term exposure (such as spray drift), automatic samplers could either be programmed to take subsamples at regular intervals that are then combined into larger samples, or they could be triggered remotely by telemetry to increase the frequency of sample collection immediately before, during and after spray events. The latter approach would depend on a good working relationship with the landowner.
2. Recreate the pattern of exposure during the following year in water bodies supporting a sensitive assemblage. Also monitor control sites upstream of the stretch where pesticides

are added. Analyse as paired control and treatment sites. Exposure should be generated experimentally (for example, spray drift recreated at the exact concentration by overspraying the water body at the relevant rate). A sensitive assemblage for macroinvertebrates could most easily be found in unpolluted small, permanent, streams with a fauna rich in mayflies, stoneflies and caddis flies, as well as *Gammarus* (typically *pulex*). Sites selected should also be appropriate for assessing impacts on the autotrophic community, most suitably the periphyton of which diatoms are a significant component. The sites should be free from agricultural diffuse pollutants. Such sites could be found immediately downstream of semi-natural woodlands and extensive semi-natural grasslands.

This experimental design addresses the central aim of the project because:

- Worst case exposure and sensitive biota would be brought together, so absence of impact would imply that impact is unlikely elsewhere in the UK. Conversely, presence of impact would indicate that impacts could not be excluded elsewhere but that further work would be needed to assess relative extent.
- The only difference between control and exposed sites would be the pesticide, so any impact could be clearly attributed; analysis of paired sites would be the most robust statistical approach.
- Prior monitoring of exposure in a high exposure situation would ensure that exposure at the test sites is realistic. This would be a stronger basis for experimental treatment than determining exposure on the basis of predictions from models.
- The timing of exposure would be known (recreated from monitoring in the high exposure situation during the previous season), so measurements to detect an effect could be deployed effectively. This would mean that the study could make use of tools such as in situ bioassays that may be difficult to deploy on an ad hoc basis. Recreation of the high exposure situation would minimise the impact of local conditions (such as farmer behaviour, wind direction) on the success of the study.

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Glossary of terms

All water – This category of results combines all water classes into a single distribution of results. This result treats all stream segments the same regardless of their location, and does not reflect the total volume of those water bodies.

Buffer – The non-crop area located between a stream segment and crop found within 20 m of the stream segment perimeter.

Buffered perimeter – The portion of a stream segment perimeter that is part of a buffer. In other words, the buffered perimeter is not directly adjacent to crop, but has crop within 20 m.

Direct adjacency – The portion of a stream segment perimeter where crop is considered to be directly adjacent to the stream segment (distance of one metre or less). This situation excludes the presence of a buffer, as there is no area between the crop and water.

Land cover classification – A data set derived from remotely sensed data (satellite imagery) that has been categorized into discrete classes based on the material present on the surface.

Line – A form of vector data that is a series of connected points. A line has a length attribute, but has no width or area. If the line data is georeferenced to map coordinates, the length value is in map units (such as metres).

Mitigated perimeter – The portion of (subset of) the buffered perimeter that has at least one metre of mitigating land cover between crop and the stream segment.

Mitigating land cover – Used in the refined potential for exposure analyses. This includes the 'trees: forest', 'trees: hedgerow', and 'scrub' land cover classes.

Non-cropped perimeter – The portion of the stream segment perimeter that is neither directly adjacent to crop, nor part of a buffer (crop is greater than 20 m away from the segment edge).

Perimeter composition – Categories of stream segment perimeter composed of direct adjacency, buffered perimeter, and non-cropped perimeter (see Figure 2.6).

Polygon – A form of vector data that is a series of connected lines returning to the starting point. A polygon has an area and perimeter attribute. If the polygon data is georeferenced to map coordinates, the area and perimeter values are in map units (such as hectares, metres).

Stream segment – Any section of a stream between other stream confluences or reaches that is within 20 m of orchards. Presence of streams in the landscape is indicated by the occurrences of hydrology linework from OS Mastermap® data.

List of abbreviations

CSL	Central Science Laboratory
Defra	Department for Environment, Food and Rural Affairs
EpiF	International workshop on Effects of Pesticides In the Field
GIS	Geographical information system
LCM 2000	Land cover map 2000
OS	Ordnance Survey
PSD	Pesticides Safety Directorate
US EPA	United States Environmental Protection Agency

Appendix 1: Electronic deliverables from risk mapping

Spatial data will be delivered in the form of separate ArcMap projects for the national- and regional-level risk mapping.

National Risk Mapping

Source Data

Catchments:
Water Framework Directive River Catchments - Approximately 6,600 catchments in England and Wales, derived from 50 m elevation data and corrected to 1:50,000 scale hydrology from the Centre for Ecology and Hydrology (Environment Agency, 2005).

Aquatic landscapes:
Hydrogeological landscape classes define ten categories of landscape classes related to aquatic ecosystems. Aquatic landscape data were developed under Defra Project PN0931. Coverage exists for all of England, Scotland and Wales (Brown *et al.*, 2006).

Regions:
Six general regions of England and Wales.

NUTS3 administrative boundaries:
These are the county administrative boundaries in vector format.

Ward-level 2003 cropping statistics:
This layer is a spatial linkage of ward administrative units to the ward-level crop statistics from the 2003 census.

Land Cover Map 2000:
Twenty-five metre resolution land cover data developed by the Centre for Ecology and Hydrology are available across the UK. The LCM 2000 dataset is a classification of broad land cover types that enables the user to determine where within a ward agricultural production is present, and how to allocate ward-level crop statistics to each catchment. (Land Cover Map 2000, CEH, © Crown copyright). Land cover classes used from the LCM 2000 data were arable cereals, arable horticulture and arable non-rotational, since all crops of interest could be assigned to one of these three classifications.

Toxic unit tables:
The raw toxic unit tables giving theoretical toxic units grouped by aquatic landscape, region, and crop class. Toxic units were given both as a combination of drift and drainage exposure, and as separate toxic units tables for drift and drainage. The toxic units were summarised considering all compounds, and separately by considering only the compounds that have not been withdrawn.

Derived Data

Intersection of catchments, aquatic landscapes, and wards:
The union of these three datasets was a key derived layer used for modifying the LCM 2000 land cover areas in catchment / aquatic landscape / ward concurrences to the crop ratio of each crop per ward.

Catchment-level toxic unit tables:
Estimated toxic unit values by catchment, joined to catchment spatial data.

Regional Risk Mapping

Source Data

OS MasterMap Hydrography & OS MasterMap Land Cover:
OS MasterMap land cover data are given in vector format. The DESCTERM field specifies land cover types either as a homogeneous land cover, or as consociations of multiple land covers. Polygons having the DESCTERM field populated with ‘orchards’ or any mixture of orchards and other land cover were used in the site selection process. Polygons and lines with the DESCTERM field as ‘inland water’ were used as hydrographic data in the site selection process and to indicate the presence of water when digitising surface water from the aerial imagery.

OS MasterMap topographic data is sourced from our topological large scale dataset. The data are captured and maintained primarily by the following three processes:
• continuous revision by our network of field offices around the country;
• centralised activity driven by external intelligence sources;
• cyclic revision by photogrammetric surveys.

OS MasterMap Aerial Imagery:
Aerial imagery used in the analyses has a ground resolution of 25 centimeters. The imagery comes in tiles or orthorectified images, created as a seamless coverage. Available imagery is acquired no earlier than 1999. OS aerial imagery was used to classify detailed land cover and hydrology in the regional risk mapping phase of the study.

Derived Data

Classified hydrology:
The classified hydrology is the vector dataset derived by digitising surface water from the OS MasterMap aerial imagery. The classified hydrology was used in conjunction with the classified land cover in regional risk mapping to characterise buffers between water and orchards.

Classified land cover:
Classified land cover is the vector representation of land cover types in the 20 m margin around the classified hydrology. Valid land cover types included orchards, grass, forest, hedgerows, scrub, bare ground, urban/developed, arable crop and other crop.

Sample points:
Points around stream segments are the locations from which measurements are made to crop.

Measurements:
The actual measurements made from stream segment perimeter to crop will be provided. Each measurement is 20 m long and identifies distance to crop, as well as land cover intersected in the buffered area before encountering crop. Only measurements encountering crop will be reported.

Metrics mapped at the stream segment level:

- buffer widths;
- transect lengths and counts through mitigating land cover;
- buffer composition.

Complete and selected grid cells:
A 5 x 5 km lattice was created as part of a systematic site selection approach in which OS MasterMap orchard polygons within 20 m of water were quantified per cell. Grid cells were subsequently ranked to determine the five cells per site with the highest areas of orchard production within a 20 m distance to water.

Appendix 2: Contribution of different regions and landscapes to risk

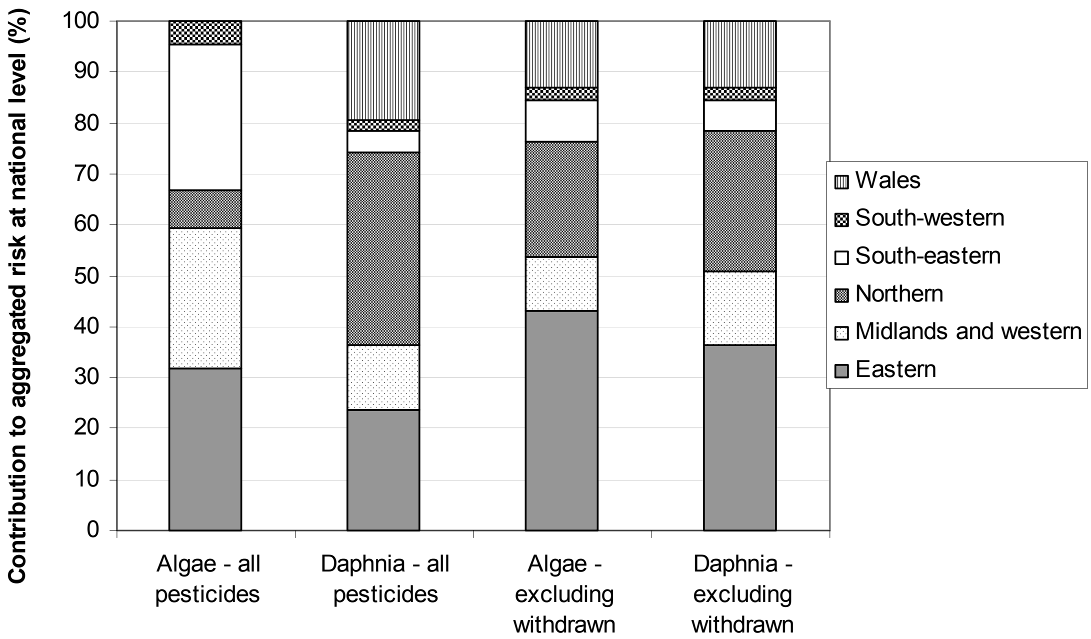

Figure A2-1. Relative contribution of the different regions to the aggregated risk shown in Figure 3.7 (note that calculations are available for Wales even though these could not be mapped due to lack of spatial cropping data)

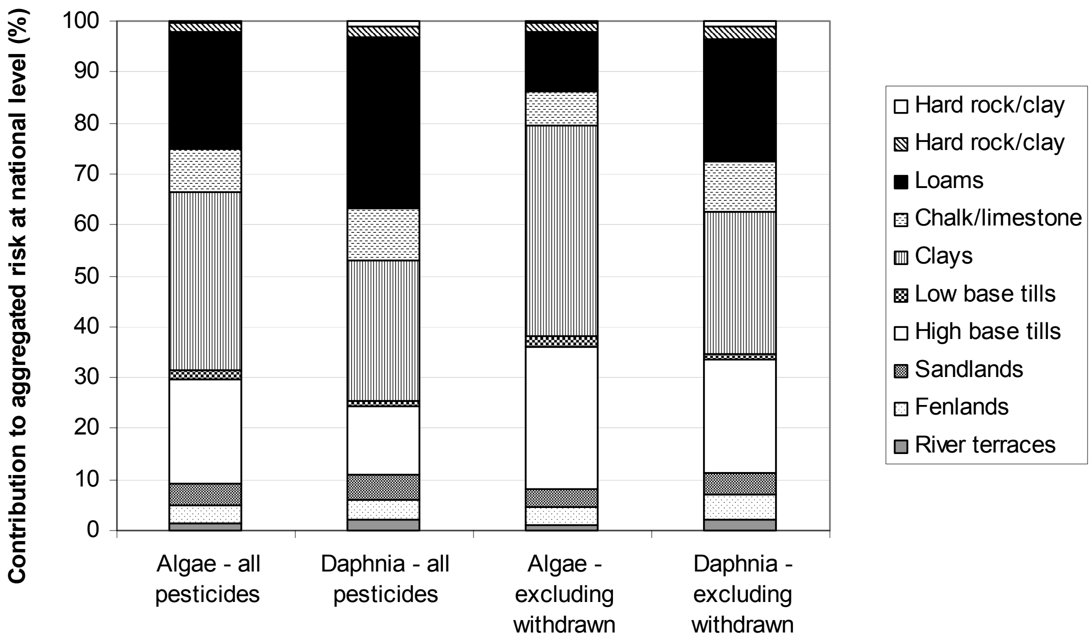

Figure A2-2. Relative contribution of the different landscape classes to the aggregated risk shown in Figure 3.7.

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Environment Agency
Rio House
Waterside Drive, Aztec West
Almondsbury, Bristol BS32 4UD
Tel: 0870 8506506
Email: enquiries@environment-agency.gov.uk
www.environment-agency.gov.uk