DEPARTMENT for ENVIRONMENT, FOOD and RURAL AFFAIRS **CSG 15**

Research and Development

Final Project Report

(Not to be used for LINK projects)

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Project title	Aquatic ecosystems in the UK agricultural landscape
DEFRA project code	**PN0931**
Contractor organisation and location	The Ponds Conservation Trust: Policy & Research, c/o Oxford Brookes University, Gipsy Lane, Headington, Oxford, OX3 0BP.
Total DEFRA project costs	£217,851
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Executive summary (maximum 2 sides A4)

1. Aquatic habitats in many parts of Europe are exposed to pesticides used to control agricultural pests and diseases. Regulatory procedures aim to limit the risk to these habitats by testing and licensing agrochemicals and restricting their application. The effectiveness of regulatory mechanisms is, however, limited by lack of biological and physico-chemical information describing the aquatic ecosystems most at risk from agrochemicals.

2. This project analysed existing national freshwater data-sets to create a characterisation of aquatic habitats in the UK agricultural landscape. Regional field data were collected to support and test the findings. Desk studies were undertaken to review the main factors that determine the exposure of, and risk posed to, aquatic species and habitats in agricultural areas.

3. A classification of agricultural landscapes was developed for England, Scotland and Wales based on soil parent material and soil hydrology. Twelve agricultural landscapes were defined, capturing differences in hydrogeology, soils, topography and cropping. A thirteenth landscape, non-agricultural land, comprised all areas unlikely to receive significant agricultural inputs of pesticide. Definitions were developed for the five main waterbody types considered in the project: rivers, streams, ponds, lakes and ditches.

4. The density of different waterbodies in the landscape classes was calculated using Countryside Survey 2000 data and a digital analysis of river length. For example, the length of non-road ditches in the different landscapes varied between 0.3 and 5.7 km/km^2. Pond densities varied between 0.6 and 3.6 ponds/km^2. Since 1990 there has been little change in either the total length of ditches, streams and rivers, or in the number of ponds and lakes in agricultural landscapes.

5. Macroinvertebrate and wetland plant assemblages were described for each waterbody type in the 13 landscape classes. The most comprehensive data were available for streams and ponds, with more limited data from rivers, ditches and lakes. Rivers supported the richest invertebrate assemblages at site level, but at the landscape level all waterbody types, including ditches, contributed similar proportions of species to national diversity. There were significant differences in invertebrate and plant species richness between agricultural landscape types.

6. An investigation was made of the proportion of invertebrate species in different waterbody types that are more sensitive to pesticides than the standard invertebrate test species, *Daphnia*. This showed that sensitive groups (including stoneflies, mayflies and amphipod crustaceans) make up a relatively high proportion of the species in rivers and streams (20% - 40%) with lower proportions in ponds and ditches (5% - 10%).

7. A relational database was developed as a repository for the data collated during the project. This comprises a MS Access database linked to a graphical user interface programmed in Visual Basic. The application displays maps and summary information on landscapes and allows user-specified interrogations to extract data from the database. The software will be distributed to interested parties free of charge via the internet.

8. A review of factors influencing the ability of species to recover from pesticide impacts showed that recovery processes operate at a variety of scales: macro (waterbody and landscape), meso (habitat) and micro (population). At macro-scale, running waters and lakes have the potential to recover relatively rapidly by recolonisation from unaffected areas of the same waterbody. Smaller and more isolated habitats such as springs, headwaters, ponds, and upper catchment ditches, are at greater risk because they often rely on more unpredictable colonisation from other waterbodies in the surrounds. At smaller scales, species recovery is affected by a multitude of variables associated with habitat type and population dynamics (birth rate, predation, disease etc.). The complex interactions between these variables means that attempts to make accurate predictions of recovery rates for specific populations are likely to prove challenging.

9. The potential use of project data in probabilistic risk assessment (PRA) was investigated. The aggregated data provided in the relational database are summarised as mean, median, standard deviation and range values (numerical values) and as proportions of sites within specified categories (categorical data). Both data types are suitable for defining probability distribution functions for use in PRA. A probabilistic calculation was undertaken for aquatic exposure via spray drift using distributions in landscape features including waterbody dimension and vegetation type for the zone between sprayed crop and bank top. For a single landscape, the standard exposure estimate was more conservative than >98% of scenarios on a cumulative distribution curve.

10. There is interest in the use of GIS to support risk assessment for pesticides and to support post-regulatory management of risk. The map of landscape classes developed for the current project could be made available as a digital (polygon) dataset. All remaining outputs from the project are derived (aggregated) data that are not spatially referenced. However, spatially referenced raw data underpin the aggregation and, subject to licensing arrangements with the originators, could potentially be manipulated for use in GIS.

11. In conclusion, data collated by the project and made available in the project database provide an ecological resource that will support and inform the future development of the pesticide risk assessment process in the UK. Specifically:

    - Information describing species occurrence in different waterbody types has implications for the choice of test species used in risk assessment. For example, evidence that pesticide sensitive invertebrate groups (including stoneflies, mayflies and amphipod crustaceans) comprise a significant proportion of the fauna of some freshwaters suggest that higher tier risk assessment could be improved by ensuring adequate representation of these groups in test systems.
    - Data describing the density of waterbody types suggests that although ponds and ditches are the predominant waterbodies in agricultural areas, streams and rivers are a significant component of some landscape types. This implies that there is a case for greater use of running water mesocosms in higher tier risk assessment.
    - Combining project information on the spatial distribution of water body types in landscape classes with cropping data and the use pattern of products (GAP), will facilitate the development of more realistic pesticide exposure scenarios. Data from the current project are already being used for this purpose in DEFRA project PS2304.

12. Further work is needed in the following areas:

    - Additional field data are required to adequately describe the biotic and physico-chemical characteristics of ditches in most agricultural landscape classes.
    - Specific data are required to describe the distribution of taxa that are an important component of first tier risk assessment e.g. zooplankton, Diptera and fish.
    - There is a need for the general protectiveness of existing regulatory measures to be confirmed in the field. There are currently very few field data that can be used either to corroborate laboratory and mesocosm results or to establish the relative importance of pesticide and other diffuse pollutants in aquatic habitats impacted by agricultural ecosystems.
    - Field data are also required to provide information on the recovery potential of some waterbody types, particularly isolated and still-water systems for which there are exceptionally few published data.
    - Additional work is needed to incorporate the landscape map and affiliated data into the SEISMIC software to improve development of risk assessment scenarios.

Scientific report (maximum 20 sides A4)

1. Introduction

Across Europe, freshwater habitats are potentially exposed to a range of agricultural pesticides. Regulatory procedures aim to limit the risk posed by these agrochemicals by testing and licensing pesticides and by restricting their application. To optimise regulatory mechanisms, so that they are effective but not punitive, requires considerable knowledge of the freshwater ecosystems most at risk from pesticides. In practice, however, there have been very few comparative studies of the physical parameters, chemistry or biology of freshwater habitats in any agricultural landscape type and there is, as a result, little information available describing the aquatic environments that regulations are designed to protect.

To facilitate the development of more refined procedures for the testing, regulation and licensing of pesticides there is a critical need for comparative data that describe the nature of aquatic ecosystems in different agricultural landscape types. The aim of this project has been to collate and summarise information from the major national freshwater datasets available in Britain and to use this to characterise and compare their biological, physico-chemical and morphological characteristics. These, and other, data have then been used to define the factors influencing the exposure and risk to waterbodies in agricultural landscapes in relation to pesticide usage and the potential of the ecosystems for recovery.

The results provide information that can support higher tier risk assessment and probabilistic approaches and will assist DEFRA's continued development of risk assessment methodologies in pesticide regulation in accordance with the Plant Protection Products Directive (91/414/EEC) and its associated Annexes.

1.1. Objectives

The project had seven major objectives. These were to:

1. Describe the spatial and temporal distribution of aquatic habitats in the UK agricultural landscape.
2. Characterise the biological, physico-chemical and morphological characteristics of aquatic systems using both existing and trial datasets, in relation to the agricultural landscape.
3. Identify the factors influencing the magnitude and duration of exposure to pesticides.
4. Identify factors which might influence the potential for ecosystems to recover from pesticide related effects.
5. Investigate the potential for the use of this information in probabilistic risk assessments.
6. Investigate the potential for use of spatially explicit tools such as GIS.
7. Identify the regulatory implications for pesticide risk assessment and risk management in the UK and make recommendations for further research.

2. Methods

2.1. Definition and distribution of waterbody types

Working definitions were developed for the five waterbody types assessed within the project: ditches, ponds, streams, rivers and lakes. Definitions were based on hydrological, morphological and biological criteria, particularly considering: (i) the range of existing definitions in common usage; (ii) practical constraints imposed by pre-existing datasets analysed in the project; and (iii) criteria that could be derived or calculated from Ordnance Survey maps.

Landscape classes were defined for the project to capture broad differences in types, properties and abundance of waterbodies, potential for exposure to pesticides (i.e. agricultural land use) and routes of movement of water (and thus potentially pesticide) from agricultural fields to water. First, the extent to which hydrogeology, soils, topography and cropping patterns co-vary across the landscape was assessed visually using the legend attributes from the 1:250,000 soil maps of England, Wales and Scotland (Mackney *et al*. 1983, MISR 1984). Next, descriptions of landscapes were set out using broad types of soil parent material as a link between topography and hydrogeology (expressed as the likelihood of presence of different types of waterbody) and including elements of the Hydrology of Soil Types classification (Lilly *et al.*1998). A digital dataset was generated using the national soil maps of England and Wales and of Scotland (both polygon datasets at scales of 1:250,000). Non-agricultural areas (defined as those unlikely to receive significant agricultural inputs of pesticide) were identified by combining urban and inland water polygons with all soil association map units with no significant agricultural usage given on the map legend. All remaining soil associations were assigned to one of 12 agricultural landscape classes using soil parent material as the classifier. Digitised boundaries for landscape classes were generated from soil association linework. The resulting map was rather fragmented where soil parent material is locally heterogeneous. A smoothed map was generated by merging small polygons wholly contained within larger polygons and by removing long, thin polygons with a resolution of *ca*. 500 m.

The spatial distribution of waterbody types was described using a variety of datasets. The length of river within each landscape class was estimated using the ESRI ArcView GIS software. Two databases were used: (i) the polygon *shape* data file for the aquatic landscape classes; and (ii) the line layer '*River*' data files from the Ordnance Survey "STRATEGI" dataset. The six river line layer files were updated to create a single river line file which was then clipped within each of the polygons for the landscape classes to produce a separate shape file for each landscape class. Each of these shape files was then converted to an arc coverage and the length queried to derive an accurate river length estimate within each landscape class. For streams, ditches and ponds, data were derived from the Countryside Survey 2000 (CS2000). The 569 1 km squares of the Countryside Survey were reclassified in terms of the 13 landscape classes of the current project. For each landscape class the mean length of ditch and stream per km^2, and the mean density of ponds per km^2, was calculated. Information from the Countryside Survey was used to describe the temporal distribution of aquatic habitats and to identify numerical and distributional changes for each major aquatic habitat type over the period 1990-2000.

2.2. Collation and analysis of national freshwater ecological datasets

Available national and regional datasets describing aquatic macrophyte and aquatic macroinvertebrate assemblages were collated for the waterbody types considered in the project (Annex 1). To facilitate direct comparisons, species lists were harmonised as far as possible. This was straightforward for the macroinvertebrate datasets that were collected using comparable methods (three-minute hand-net samples). The following groups were included in the analysis: flatworms (Tricladida), water snails and bivalves (Gastropoda and Bivalvia, excluding *Pisidium* spp), leeches (Hirudinea), shrimps, slaters and crayfish (Malacostraca), mayflies (Ephemeroptera), stoneflies (Plecoptera), dragonflies (Odonata), water bugs (Heteroptera), water beetles (Coleoptera), alderflies (Megaloptera) and caddis flies (Trichoptera). Only taxa that had been identified to species level were included. Diptera and worms (Oligochaeta) were excluded from the analysis because information on these groups was not available from all waterbody types. All comparisons were based on single samples collected in summer (June, July, August). Plant survey methods were more varied (e.g. collected over different survey lengths in different waterbody types), and differed considerably in the range of species included. To maximise compatibility between plant lists, a standard species list was compiled. This omitted bryophytes and charophytes because these were not recorded in all surveys. The number of plant hybrids also varied considerably between surveys, probably reflecting the taxonomic skills of the recorders. To address this, hybrid plants were omitted from species richness calculations unless the parents were absent from the site. The plant and invertebrate datasets were analysed to: a) derive mean and range biodiversity values (e.g. species richness, species rarity) for each waterbody type, b) evaluate the alpha and gamma diversity characteristics of aquatic habitats in the agricultural landscape, and c) describe the composition of community types for each freshwater habitat including the occurrence of sensitive taxa. In these analyses, rarity was assessed using a Species Rarity Index. This was based on the average rarity value of species at each site using the following scoring system: 1=common species, 2=local species, 4=Nationally Scarce, 8=Red Data Book (RDB) Near Threatened, 16=RDB Vulnerable, 32=RDB Endangered.

For each of the available datasets, the physico-chemical and morphological features of waterbody types were described for each agricultural landscape class. This included, where available, assessment of mean and range values for attributes relevant to pesticide risk assessment (e.g. waterbody size, morphology, flow characteristics, pH, permanence, sediment characteristics, abundance of aquatic vegetation, bankside vegetation, distance to crop) and ecosystem driving variables (e.g. nutrient status, substrate composition).

2.3 Collection and analysis of regional freshwater and ditch datasets

Additional field data were collected for the project to address two issues: (i) lack of adequate physico-chemical and biological information describing ditches in agricultural landscapes; and (ii) the need to validate the findings of the meta-analyses of national and regional datasets (2.2 above). New ditch data were gathered from four field study areas in contrasting agricultural landscape classes (LCs) within a 10 km^2 area. These were: Spalding (LC2), Morpeth (LC4), Whitchurch (LC5), and Kington (LC7). From each area, biological and physico-chemical field data were collected from 10 randomly located ditch sites. Two regional datasets were used to validate trends evident in the national data. Both were collected using directly compatible methods from waterbody types in contrasting, representative, landscape classes from a 100 km^2 area. The first dataset (80 sites) was from a pre-existing study centred on Coleshill (Oxfordshire) and encompassed the Brimstone Farm study area (Williams *et al.* in press). A second regional trial dataset (39 sites) was collected from the Whitchurch area of Shropshire.

2.4 Creation of relational database

A relational database was constructed as a repository for the processed (aggregated) data on the properties of the landscape classes. Data tables were imported from into a single MS Access database comprising 22 tables. A graphical user interface (GUI) was written in Visual Basic to allow users to display information, interrogate the database and extract data from the database into comma separated value files. Within the GUI, the ESRI MapObjectsLT software library was used to enable the display and interrogate the map of landscape classes. The database and GUI are Windows-based software designed for use with either Windows 2000 or Windows XP platforms.

Results

3. Spatial and temporal distribution of aquatic habitats in the agricultural landscape

3.1 Identification of landscape types potentially exposed to pesticides

A total of 12 agricultural landscape classes, plus a 13th landscape class, non-agricultural land, were identified for England, Scotland and Wales. The landscape classes are summarised in Table 1 and their spatial distribution shown in Figure 1.

3.2 Definition of aquatic habitat types

Working definitions of the five waterbody types assessed during the project (ditches, ponds, streams, rivers, lakes) were developed based on hydrological and morphological criteria (Table 2).

The biological validity of this classification was assessed using a pre-existing regional dataset that compared the wetland plant and aquatic macroinvertebrate assemblages of ponds, ditches, streams and rivers (Williams *et al.* in press). Multivariate analysis of the aquatic invertebrate data generally supported the classification. Analysis of plant data suggested considerable similarity between the plant assemblages of the different waterbody types.

3.3 Ecological data search

The major national UK ecological datasets for freshwater habitats are well known. However, an additional check was made to ensure that useful smaller or regional studies had not been overlooked. A particular effort was made to locate information for ditches where no national survey data are available. A large number of local ditch surveys have been undertaken, but the vast majority are from fenland areas (Landscape Class 2). In addition, a wide range of survey methodologies have been used making it difficult to compare results. For the current analysis we mainly collated ditch datasets that were accessible electronically. However, where information was exceptionally scarce, hard copy data were digitised. A list of the datasets used in the study is given in Annex 1.

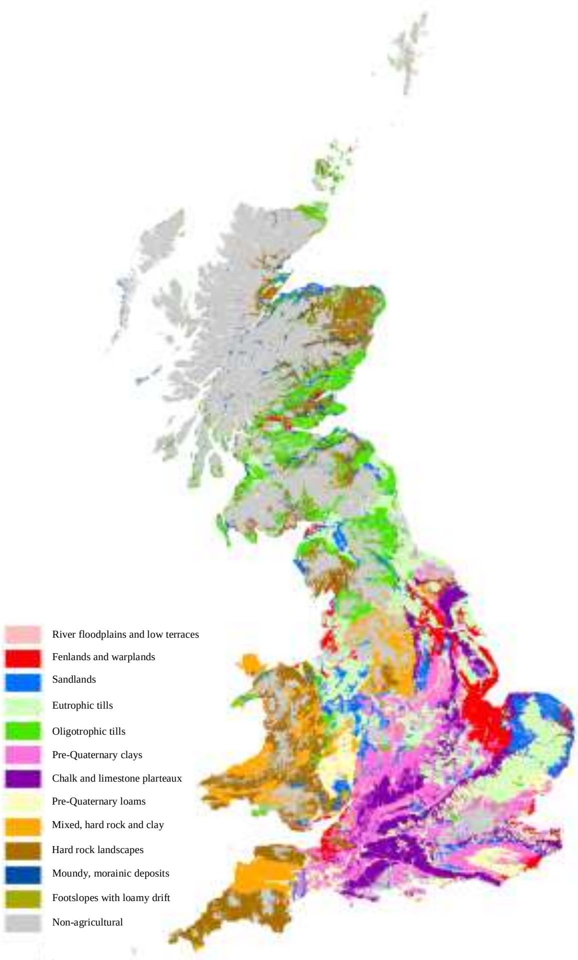

Figure 1. Distribution of landscape classes in England, Scotland and Wales

Table 1. Summary of the landscape classes derived for England, Scotland and Wales

No.	Landscape	Description	Total area (km^2)	Crops (minor crops in parentheses)
1	River floodplains and low terraces	Level to very gently sloping river floodplains and low terraces	7,781	Permanent grass, some cereals and oil-seed rape, probably more intensive on terraces
2	Warplands, fenlands and associated low terraces	Level, broad 'flats' with alluvial very fine sands, silts, clays and peats	9,017	Cereals (oil-seed rape, beans), sugar beet, potatoes, peas, vegetables, top fruit
3	Sandlands	Level to moderately sloping, rolling hills and broad terraces. Sands and light loams	10,871	Cereals (oil-seed rape, beans and peas), sugar beet, potatoes (peas in East Anglia)
4	Till landscapes	Level to gently sloping glacial till plains. Medium loams, clays and chalky clays, with high base status (eutrophic). Some lighter textured soils on outwash	22,151	Cereals, oil-seed rape and beans (peas in E. Yorks.), permanent and rotational grass (mainly in west)
5	Till landscapes	Level to gently sloping glacial till plains. Medium loams and clays with low base status (oligotrophic). Some lighter textured soils on outwash	15,449	Permanent and rotational grass with some cereals and oil-seed rape
6	Pre-quaternary clay landscapes	Level to gently sloping vales. Slowly permeable, clays (often calcareous) and heavy loams. High base status (Eutrophic)	19,706	Permanent grass, cereals (>10-15%), leys, oil-seed rape, maize (not in NE or Weald) and beans
7	Chalk and limestone plateaux and coombe valleys	Rolling 'Wolds' & plateaux with 'dry' valleys; shallow to moderately deep loams over chalk & limestone	14,197	Cereals (and oil-seed rape, beans), sugar beet, potatoes, peas
8	Pre-quaternary loam landscapes	Gently to moderately sloping ridges, vales and plateaux. Deep, free-draining & moderately permeable silts & loams	10,072	Permanent & rotational grass, cereals and oil-seed rape with some beans, grass, hops and fruit
9	Mixed, hard, fissured rock and clay landscapes	Gently to moderately sloping hills, ridges and vales. Moderately deep free draining loams mixed with heavy loams and clays in vales	12,259	Permanent grass, rotational grass and some cereals (<10-15%)
10	Hard rock landscapes	Gently to moderately sloping hills and valleys. Moderately deep free draining loams over hard rocks. Some slowly permeable heavy loams on lower slopes and valleys	23,342	Permanent grass, rotational grass and some cereals (<10-15%)
11	Moundy morainic and fluvioglacial deposits	Gently and moderately sloping mounds, some terraces. Free draining morains, gravels & sands on mounds, poorly draining gleys in hollows	2,270	Permanent and rotational grass, some cereals
12	Footslopes with loamy drift	Concave slopes or depressional sites often with springlines	1,081	Permanent and rotational grass
13	Non-agricultural	All areas not cultivated with arable (including orchards, soft fruit and horticultural) or maintained grassland	79,690	No crops
Total	-	-	227,886	-

Table 2. Definitions aquatic habitats used in the study

Waterbody	Definition
Lakes	A body of water greater than 2 ha in area (Moss *et al*. 1996). Includes reservoirs and gravel pits of sufficient size.
Ponds	Waterbodies between 25 m^2 and 2 ha in area which may be permanent or seasonal (Collinson *et al.* 1995). Includes both man-made and natural waterbodies.
Ditches	Man-made channels created primarily for agricultural purposes, and which usually: (i) have a linear planform; (ii) follow linear field boundaries, often turning at right angles; and (iii) show little relationship with natural landscape contours.
Streams	Small lotic waterbodies created mainly by natural processes. Marked as a single blue line on 1:25,000 Ordnance Survey (OS) maps and defined by the OS as being less than 8.25 m in width. Stream differ from ditches by: (i) usually having a sinuous planform; (ii) not following field boundaries, or if they do, pre-dating boundary creation; and (iii) showing a relationship with natural landscape contours e.g. running down valleys.
Rivers	Larger lotic waterbodies, created mainly by natural processes. Marked as a double blue line on 1:25,000 OS maps and defined by the OS as greater than 8.25 m in width.

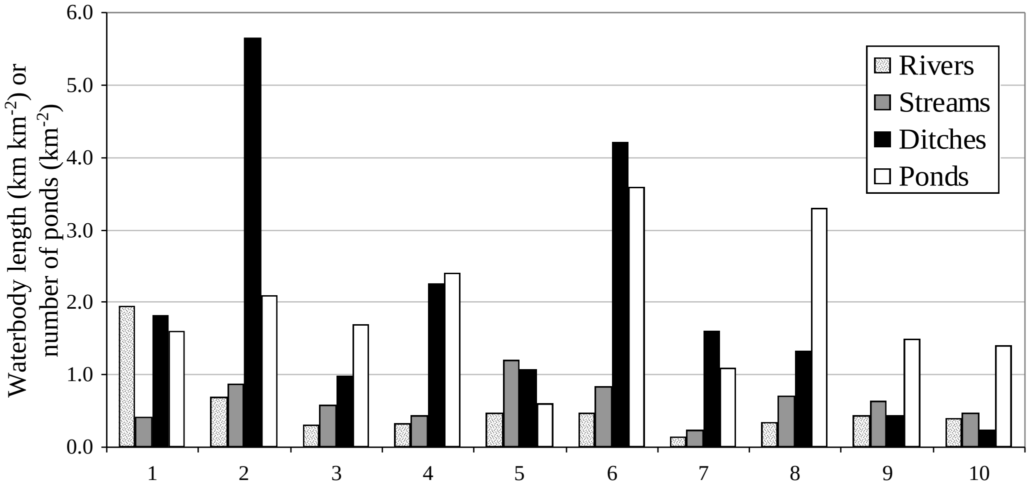

Figure 2. Density of waterbodies in agricultural landscape classes

3.4 The spatial and temporal distribution of different types of aquatic habitat

The mean density of rivers, streams, ponds and ditches were calculated for landscape classes 1-10 using the methods described in Section 2.1. Waterbody length/density estimates were not made for non-agricultural areas (LC 13), or for landscapes 11 and 12 which occupy a small proportion of the landscape and are relatively non-agricultural. The results of this analysis are summarised in Figure 2. Waterbody density data within different landscape classes was then combined with information on land use and field sizes to generate a series of idealised schematic maps for landscapes 1-10. The schematic maps give an impression of the amount of water and arable or grassland cultivation in different landscapes and of how the two features are likely to inter-relate. For example, the analysis showed that almost every side of fields in Landscape 2 (fenlands) is likely to be bordered by a ditch. Some expert knowledge was captured within the schematic maps regarding the extent to which the different waterbodies are likely to occur within arable, grassland or non-agricultural areas. All ten schematic maps are provided in Annex 2 together with representative photographs of the landscapes.

Temporal change data from the Countryside Survey indicate that the area of streams, rivers, ditches and lakes did not change significantly between 1990 and 2000. There was a small increase in pond numbers during this time, reversing a long period of decline over the previous 50 years (Haines-Young *et al.* 2000).

4. The characteristics of aquatic systems in the agricultural landscape

4.1 Biological characteristics

4.1.1 Available data

The available national biological datasets were analysed for five waterbody types (streams, rivers, ponds, lakes and ditches), and two biotic groups: (i) macroinvertebrates; and (ii) macrophytes, including submerged aquatic, floating-leaved and emergent plants.

For both invertebrates and plants, moderately comprehensive datasets were available for streams, ponds and, to a lesser extent, rivers. For lakes, data were available for macrophytes but not for invertebrates. Ditch data were particularly poorly represented for all landscape classes except fens and river valleys. Field data gathered for the project were used to partly supplement the ditch dataset but reliable descriptions of assemblages was possible for only 5 of the 13 landscape types. For macroinvertebrates, all data were gathered using a similar methodology (3 minute hand-net samples), so results from the different waterbody types can be directly compared. Macrophyte datasets were, however, collected using different methodologies. This prevents between waterbody type comparisons of real numbers (e.g. species richness), although indices and proportional values are probably broadly comparable.

4.1.2 Results

The relational database that accompanies this report provides mean and range values for the species richness and rarity indices of each landscape class, in each waterbody type. It also shows the proportion of sites in which each plant and invertebrate species was found. Further analysis of these data, organised by waterbody type, is given in Annex 3. The main findings are summarised below.

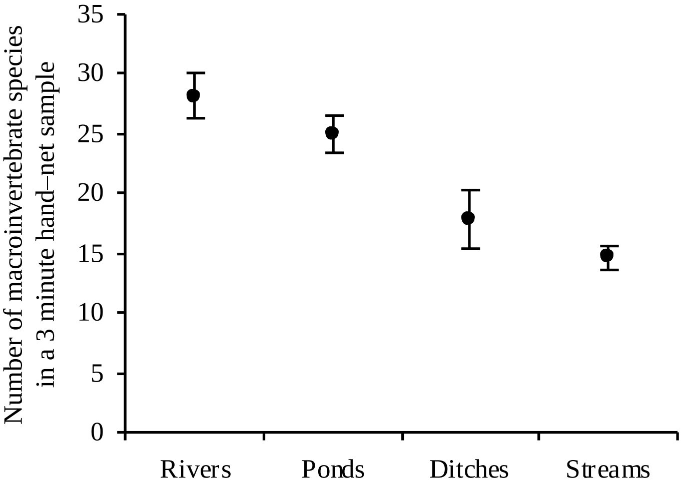

Figure 3. Macroinvertebrate species richness in aquatic habitats in the agricultural landscape. Error bars are 95% confidence limits.

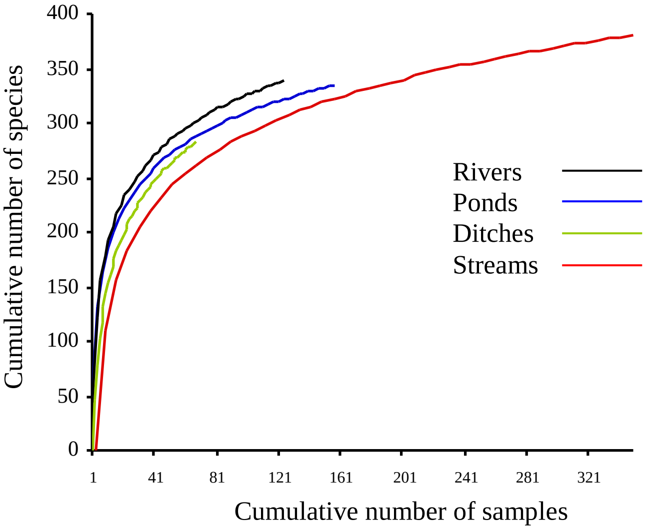

Figure. 4. Macroinvertebrate gamma diversity in four waterbody types in the agricultural landscape.

4.1.2.1 <u>Macroinvertebrates</u>

Site richness: Overall, site richness (alpha diversity) was greatest in rivers (mean of 28.2 species per 3 minute sample), followed by ponds (25.0 species), ditches (17.8) and streams (14.6) (Figure 3). For rivers, ponds and ditches mean species richness differed significantly between landscape classes, although there was no consistent pattern in landscape richness. Thus in rivers and ditches the richest assemblages were found in landscape classes 1 and 2 whereas for ponds, all lowland landscapes had similar numbers of species, but lower richness in the two upland landscape classes 10 and 13.

Gamma diversity: Gamma diversity is a measure of the total number of species found in each waterbody type. It is a useful supplement to the measure of alpha diversity because it is possible for sites that are individually species poor to be collectively variable, so that the waterbody type as a whole is species rich at regional or national level. In practice, comparing waterbody gamma diversity is difficult if, as in the current case, there are unequal numbers of sites for each waterbody type. Figure 4, therefore, presents the data as accumulation curves. These suggest, rather surprisingly, that nationally all four waterbody types support a similar total number of species. This is interesting because it suggests that ditches, which are often considered species-poor habitats may, as a whole, support a considerable proportion of the aquatic biodiversity present in agricultural environments.

Rarity: The average rarity value of each site for invertebrate species was calculated using a Species Rarity Index (SRI, see Section 2.2). The results showed that ponds supported significantly more uncommon species than other waterbody types, with rivers, ditches and streams supporting similar, but lower, proportions of rare species. There was relatively little variation in SRI values between landscapes (Table 3). Only ditches showed significant between-landscape differences, with LC 2 (fens and warplands) supporting a higher proportion of rare species than the other landscapes for which data were available.

Assemblage composition: For each waterbody type and landscape class, invertebrate assemblages were divided up by major taxonomic group (proportions of bugs, beetles, shrimps, mayflies etc.). The results indicate that invertebrate composition shows significant differences between waterbody type. Flowing waters (streams and rivers) are particularly rich in caddis, mayflies and stoneflies whilst ponds are dominated by beetles and bugs. Ditches are somewhat intermediate (Table 4).

Table 3. Species Rarity Index values for aquatic habitats in the agricultural landscape.

		Landscape class												
	All	**1**	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**	**10**	**11**	**12**	**13**
Ponds	**1.15**	1.12	n/d	1.25	1.21	n/d	1.15	1.11	1.12	1.10	1.01	n/d	n/d	1.14
Rivers	**1.07**	1.06	n/d	1.04	1.05	n/d	n/d	n/d	n/d	n/d	1.18	n/d	n/d	1.05
Streams	**1.09**	1.07	1.15	1.05	1.04	1.16	1.05	n/d	1.12	1.05	1.11	1.09	n/d	1.08
Ditches	**1.07**	n/d	1.14	n/d	1.06	1.05	1.09	n/d	1.01	n/d	n/d	n/d	n/d	n/d

n/d: no data available from that landscape. Ponds have significantly more uncommon species than other waterbody types (F = 6.587, df = 3, 676, p < 0.001).

Table 4. Contribution of major macroinvertebrate groups to species richness in streams, rivers, ditches and ponds

Waterbody type	Flatworms	Snails	Bivalves	Leeches	Crustaceans	Mayflies	Stoneflies	Dragonflies	Water bugs	Water beetles	Alder flies	Caddis flies
Streams (n = 348)	2.8%	9.4%	0.3%	5.1%	6.1%	12.8%	15.9%	1.8%	2.4%	18.6%	1.2%	23.5%
Rivers (n = 124)	2.4%	16.1%	1.5%	6.7%	5.1%	16.5%	9.1%	2.0%	2.4%	16.0%	1.3%	20.6%
Ditches (n = 68)	1.7%	14.5%	0.3%	6.4%	8.4%	2.6%	1.3%	2.1%	6.9%	41.5%	1.8%	12.4%
Ponds (n = 157)	1.9%	11.0%	0.8%	7.4%	5.0%	2.9%	0.1%	5.9%	17.9%	42.6%	1.3%	3.5%

Based on the most complete datasets (streams and ponds), there seems to be considerably greater variation in the faunal composition of running waters than of still. Thus the contribution to the fauna made by each major taxonomic group in ponds remains similar across all landscape types. For streams, however, taxonomic composition changes broadly in line with landscape acidity, from being largely dominated by insects, particularly stoneflies, in more acid areas (mainly higher landscape class numbers), to supporting a higher proportion of more sedentary taxa (snails, leeches, crustaceans) in more eutrophic agricultural landscapes.

Sensitivity: the sensitivity of species to pesticide impacts was evaluated by comparing the number and proportion of species present that were less or more sensitive to pesticides than *Daphnia* using the index developed by Wogram and Liess (2001). The findings suggest that ditches and, particularly, ponds generally support a relatively low proportion of more sensitive taxa (stoneflies, mayflies, amphipod crustaceans) with species in these groups typically making up 5% - 10% of the total faunal richness. In streams and rivers sensitive taxa typically comprise 20% - 40% of the species. In streams there is a trend for more upland/acid landscapes to support a higher proportion of sensitive species than lowland landscapes. There is less evidence of this trend in rivers, although paucity of river data makes this difficult to determine. In ponds and ditches there are few differences between the proportion of sensitive species in different landscape classes (see Annex 3).

4.1.2.2 Macrophytes

As noted above, the plant datasets from each waterbody type were collected using different methods, including different survey lengths and areas, so it is not possible to compare plant richness in the different waterbody types in terms of alpha or gamma diversity. Rarity indices and proportional distributions within landscape classes are, however, broadly comparable.

Richness of landscape classes: For streams, ponds and ditches, there is a tendency for fen and river landscapes (LC1, LC2) to have higher plant species richness than other agricultural landscapes. For rivers and lakes this trend was less apparent and mean plant species richness was greatest in landscape classes other than LC1 and LC2).

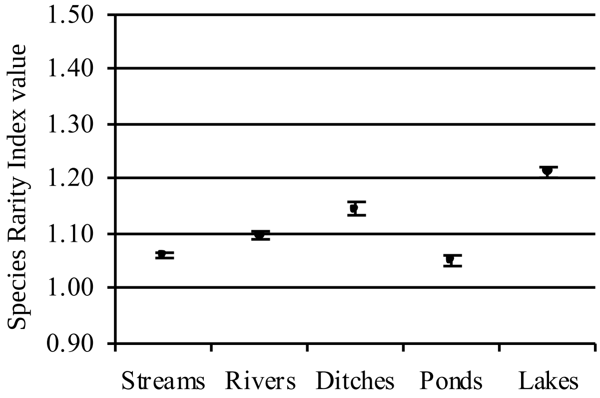

Figure 5. Mean plant Species Rarity Index values for aquatic habitats in the agricultural landscape

Rarity: On average, lakes and ditches support the highest proportion of uncommon plant taxa, followed by rivers, streams and ponds (Figure 5). This may, however, have been an artefact of the sampling strategy as most lakes (79%) were in the non-agricultural landscape (LC13) and ditch surveys were focussed on sites of known botanical richness (see also Section 4.2). Differences between waterbody types are statistically significant. Rarity index values are generally similar across the landscape classes for streams, rivers, ponds and lakes. Ditches are, however, much more varied, and support a high proportion of uncommon taxa in LCs 1 and 2. LC1 is particularly rich in uncommon plants (see Annex 3).

Assemblage composition: Plant assemblages were divided into three broad groups: (i) submerged; (ii) floating-leaved; and (iii) emergent. In all waterbody types emergent plants comprised the majority of the flora, and floating-leaved species a small minority. Comparison of the waterbody types showed that the proportion of aquatic plants is relatively low in ponds (mean = 8% of species) and highest in rivers and lakes (means = 20% and 31%, respectively). In ditches, the proportion of aquatics is variable, being high in the larger and more permanent ditches associated with fens and floodplains, and low in other landscapes, presumably because ditches are more seasonal in these areas. Ponds have, as would be expected, more floating-leaved species than other waterbody types: many floating species are not rooted so are more uncommon in running waters. Across the landscape classes, most waterbody types have a higher proportion of submerged and floating leaved species in base-rich agricultural areas, and a greater proportion of marginal species in predominantly acid non-agricultural landscapes. The main exception was the lakes which show the reverse trend (Annex 3).

Fig 6. Macroinvertebrate and macrophyte species richness in the Coleshill and Whitchurch study areas.

4.2 Targeted field work

The national data described above are not ideal in that they were collected for different purposes, over a range of time periods and, for macrophytes in particular, used different survey methods. In order to investigate the validity of results from these datasets, regional field data were collected from two areas: Whitchurch (Shropshire) and Coleshill (Oxfordshire). The survey methods used were compatible between different waterbody types so that the findings could be directly compared. In the Coleshill study area data were available from streams, rivers, ditches and ponds. In the Whitchurch area data were gathered on streams, ponds and ditches. There were no rivers in the Whitchurch study area.

Site richness: In both study areas, results confirmed trends seen in the national datasets with rivers (where present) the richest habitats, followed by ponds, with streams and ditches supporting fewer species.

Gamma diversity: The regional datasets provided a better assessment of gamma diversity patterns than the national data because sample sizes were similar for all waterbody types. The results show that ponds supported the largest proportion of species (macrophytes and macroinvertebrates) in both study areas. In the Coleshill area rivers had the next largest total number of species, with streams and ditches supporting similar, but lower, numbers of species in both areas. These results were consistent with the national trends which suggested that all habitats made a similar contribution to overall species richness.

Rarity: In both study areas ponds normally supported more uncommon plants and invertebrates than other habitat types, both at site level and for the region as a whole. This agreed with national trends for macroinvertebrates but contrasted with trends for macrophytes where lakes and ditches supported the greatest number of rare species. The latter confirms the suspicions, noted previously, that the national-scale analyses of plant data for lakes and ditches are not entirely satisfactory.

Sensitivity to pesticides. An advantage of the direct comparability of the regional studies is that they allow not only the number of sensitive taxa (e.g. species of mayflies, stoneflies, amphipod crustaceans) to be compared, but also the proportion of the total number of *individual* invertebrates in these groups. The regional results show that the number of sensitive species was broadly comparable with the national data for most waterbody types. However, because amphipod crustaceans, in particular, are often a numerous group, the proportion of individuals that were sensitive to pesticides were generally higher. In the Coleshill study area for example, within the four waterbody types for which data are available, the mean proportion of species that were sensitive to pesticides was relatively low (ponds 7.1%, rivers 13.5%, ditches 13% and streams 14.1%). The number of sensitive individuals was, however, considerably higher (ponds 39%, rivers 44%, ditches 59% and streams 61%).

4.3 The physico-chemical characteristics and morphology of aquatic habitats

4.3.1 Available data

National datasets were obtained from Environment Agency and DEFRA monitoring programmes and from surveys of rivers, streams and ponds. In contrast, ditch data were poorly represented for all landscape classes except fens and river valleys. Field data gathered for the project were used to partly supplement the ditch dataset, together with data from a previous ADAS survey of ditches in Environmentally Sensitive Areas. However, reliable descriptions of ditch characteristics were available for only 3 of the 12 landscape types. Table 5 gives the total number of data points available by landscape class. A full list of datasets used is given in Annex 1.

4.3.2 Database

The database that accompanies this report provides summary statistics for the morphological features of the defined water body types by landscape class. This includes data on channel width, water body area, depth, flow characteristics, bank angle, substrate depth and composition. Also included are bank-side features such as distance from the bank top to an arable crop, abundance of aquatic vegetation, bank scrub and tree density and adjacent land use. Up to 15 chemical parameters are represented in the database, including pH, conductivity, BOD, COD, suspended solids and nutrients. Datasets were filtered to exclude monitoring sites potentially impacted from urban or industrial situations. Although the chemical determinands reported give an indication of the range of values for water quality indicators across the landscape classes (rivers, streams and ponds) it should be recognised that the dataset is limited. Caution should be used when attempting, for example, alignment of physico-chemical properties with the biological data described above. Examples of the collated/processed data available from the project database are given in Annex 4 for rivers and ponds.

Table 5. Number sites from which data on physico-chemical and morphological properties of water bodies were available in the 13 landscapes (parentheses indicate that the water body is rarely found in that landscape).

		Number of data points			
No.	**Landscape**	**Rivers**	**Streams**	**Ponds**	**Ditches**
1	River floodplains and low terraces	2457	153	28	259
2	Warplands, fenlands and associated low terraces	679	98	11	1259
3	Sandlands	415	77	17	(5)
4	Till landscapes	642	241	32	30
5	Till landscapes	232	85	6	10
6	Pre-quaternary clay landscapes	7738	348	55	11
7	Chalk and limestone plateaux and coombe valleys	307	96	(20)	(-)
8	Pre-quaternary loam landscapes	421	319	22	9
9	Mixed, hard, fissured rock and clay landscapes	508	109	16	(-)
10	Hard rock landscapes	818	204	(14)	(-)
13	Non-agricultural	1123	432	57	8

Note: Landscape classes 11 and 12 were not analysed because too few data were available.

4.4 Aquatic habitats database

The aquatic habitats database comprises a MS Access relational database with a graphical user interface programmed in Visual Basic. The GUI can be used to display pictures comprising the map of landscape classes, representative photographs for each landscape and the schematic maps for each landscape. The map of landscape classes includes facilities to zoom in and out, to display the landscape class and grid reference at a point of interrogation and to display the distribution of a single landscape class only. Summary tabular data can also be displayed for each landscape class including: typical rotations, normal frequency of cultivation of major crops; vulnerability indices for aquatic exposure via spray drift, drainflow and surface runoff; average field size; land use expressed for landscape across the whole of England and Wales or for a specified Environment Agency region only; HOST classes present in the landscape; and densities of each type of waterbody.

The most important function of the interface is extraction of data from the database. The user is able to design reports by landscape class (all or any single class) and waterbody type (all or one specific type). The data included in the database are classified into chemical, morphological and biological information and the user can select one or more sub-categories of data from each type. Data can then be exported as a comma separated values (csv) file suitable for importing into any spreadsheet application. Extracted data may be either categorical (in which case the number of sites with properties in defined categories is reported) or numerical (here the report contains mean, median, standard deviation, minimum and maximum values for the specified property). The software is currently distributed to eight end-users for beta-testing. Once necessary modifications have been made, it is anticipated that the software will be distributed to interested parties free of charge via the internet (Pesticides Safety Directorate and/or Cranfield University).

5. Factors influencing the magnitude and duration of exposure to pesticides

5.1 Characterising patterns of pesticide usage in each agricultural land type

A spatial dataset for agricultural land use has previously been developed for DEFRA project NT2206 by combining Parish Agricultural Census data for 1995 with the Land Cover Map of Great Britain. This dataset was overlaid with the spatial dataset for landscape classes to identify cropping patterns in the different landscapes. Cropping varies significantly on an east to west axis across England and Wales, so regional analyses based on the eight Environment Agency regions were also undertaken. A small survey of crop rotations divided by region and broad soil type was undertaken for a DEFRA project on Sustainable Soil Management through Improved Tillage Practices. The results of this survey were used in combination with cropping information from soil monographs to assign two to four representative cropping rotations for each landscape. The maximum frequency of cultivation on a single field was also calculated for the major crops. The use of remotely-sensed information to enhance areas of the database was assessed but ultimately rejected (see Annex 5).

The information on cropping patterns in the database can be used with knowledge of any geographical or soil-related factors influencing use of a pesticide to estimate the distribution of use within the different landscapes. In addition, the average input of herbicides, insecticides and fungicides per unit area of each landscape (i.e. averaged over the whole area including non-agricultural) was calculated from the land use information and statistics from Pesticide Usage Surveys for the various crops. Results of this analysis are provided in Table 6. Landscape 2 (fenlands and warplands) receives the highest loading of pesticides because it is largely intensive arable land and several crops receiving high inputs of pesticide (e.g. sugar beet and potatoes) are widely cultivated. There is almost an order of magnitude difference in the total input of pesticide per unit area between the different landscapes.

Table 6. Summary of loading of pesticides for the different landscapes

	Landscape class	Average input of pesticide across whole landscape (kg/ha)			
		Herbicides	Insecticides	Fungicides	Total
1	River floodplains and low terraces	0.729	0.048	0.333	1.111
2	Warplands, fenlands and associated low terraces	1.428	0.126	0.747	2.302
3	Sandlands	0.803	0.063	0.402	1.268
4	Till landscapes (eutrophic)	0.981	0.051	0.398	1.429
5	Till landscapes (oligotrophic)	0.384	0.015	0.123	0.522
6	Pre-quaternary clay landscapes	0.756	0.044	0.308	1.108
7	Chalk and limestone plateaux and coombe valleys	1.012	0.052	0.398	1.462
8	Pre-quaternary loam landscapes	0.713	0.072	0.398	1.182
9	Mixed, hard, fissured rock and clay landscapes	0.294	0.012	0.085	0.391
10	Hard rock landscapes	0.233	0.011	0.075	0.319

5.2 Crop rotations

A small survey of crop rotations divided by region and broad soil type was undertaken for a DEFRA project on Sustainable Soil Management through Improved Tillage Practices. The example rotations were refined with reference to ADAS crop specialists, and can be regarded as typical for each region.

5.3 Changes in cropping and pesticide use

In the medium term, any changes in cropping patterns are most likely to be driven by Agenda 2000 - the on-going policy shift of the Common Agricultural Policy (CAP) away from production-based support towards environment-based payments for farmers. There have been only limited effects of this policy to date (Economic Evaluation of Agenda 2000, DEFRA 2003) with decreases in prices and changes in Arable Area Payments resulting in a reduction in area planted for some crops, especially oilseed rape, which showed a 34% reduction between 1998 and 2000. Recent changes to set-a-side obligations have also caused fluctuations in the total area of arable crops grown, with a reduction of 7% between 1998 and 2000.

Longer-term reductions in the total mass of pesticides applied to arable crops will result from the current introduction of new products active at much lower rates of application and to growers applying fungicides and herbicides at reduced rates per hectare. For example, metsulfuron-methyl, applied to around 425,000 hectares of wheat for broad-leaved weed control, was applied at an average active substance rate of 4 grams/ha in 2000. In 1990, the principal broad-leaved weed herbicide applied to wheat was mecoprop, applied at an average active substance rate of 1.93 kg/ha to 484,000 ha. The average rate of application of individual fungicide products to wheat in 2000 was 0.44 of the label recommendation (DEFRA 2000).

5.5 The relationship between pesticide use and the risks and routes of exposure

Generalised indices were developed to express the vulnerability of waterbodies in the different landscape classes to pesticide exposure via spray drift, drainflow and runoff (Table 7). It is intended that the table could be used to select appropriate assessment scenarios based on the dominant route of exposure for a pesticide. Indices range from 0 (low vulnerability) to 1 (high vulnerability) and refer to the most vulnerable waterbodies in the landscape (e.g. ditches for exposure via spray drift). Each index was derived as the product of two factors. For drainflow, this was a score for extent and types of drains and a score for extent of preferential flow within dominant soils. The index for surface runoff is generated from scores for slope and soil type (silty soils being ranked the most vulnerable). Vulnerability for spray drift was derived from scores based on density of the ditch network and relative predicted environmental concentrations from a probabilistic calculation of drift (see Section 7 for details).

Table 7. Indices to express the vulnerability of aquatic exposure to pesticides in agricultural landscapes

Landscape class		Relative vulnerability index for exposure (0-1)		
		Spray drift	Drainflow	Surface runoff
1	River floodplains	0.45	0.13	0.13
2	Fenlands and warplands	0.75	0.25	0.00
3	Sandlands	0.06	0.00	0.35
4	Eutrophic tills	0.68	0.68	0.18
5	Oligotrophic tills	0.40	0.68	0.25
6	Pre-quaternary clays	0.90	1.00	0.21
7	Chalk & limestone	0.10	0.00	0.42
8	Pre-quaternary loams	0.50	0.25	0.56
9	Hard rock with clay	0.03	0.19	0.24
10	Hard rock	0.03	0.00	0.30

5.6 Basic exposure profiles for example scenarios

Basic exposure profiles were taken from those calculated by the FOCUS surface water scenarios group (FOCUS 2001) to evaluate the vulnerability indices for exposure reported above. First, correlations between the FOCUS scenarios and the landscape classes were assessed. The following were found to correlate: D3 with landscape 3 (sandlands); D4 with landscape 5 (oligotrophic tills); D2 with landscape 6 (clays); D5 with landscape 9 (hard rock and clay); and R1 with landscape 8 (loams). Next, maximum predicted concentrations for the water phase at Step 3 were extracted from the FOCUS surface water scenarios report. Results for an autumn application of five pesticides at 100 g a.s./ha are provided in Table 8. The correspondence between PEC's in Table 8 and indices in Table 7 is not exact. The indices include several landscape factors (e.g. density of waterbodies, size of waterbodies) whereas FOCUS calculations are edge-of-field with several simplifying assumptions which reduce differences between scenarios. Nevertheless, the data in Table 7 confirm the relative vulnerability of landscape 6, the importance of runoff in landscape 8 and the relative balance between spray drift and drainflow as routes of exposure in the different landscapes.

Table 8. Maximum predicted concentrations of five pesticides for FOCUS scenarios correlating with UK landscape classes (based on autumn application at 100 g a.s./ha). The exposure route giving rise to the maximum concentration is given in parentheses.

	Compound A	Compound D	Compound E	Compound F	Compound I
Koc (ml/g)	10	10	100	1000	1000
DT50 (d)	3	30	30	30	300
D3/LC3 (µg/l)	0.50 (drift)	1.54 (drainage)	5.20 (drainage)	0.63 (drift)	0.50 (drift)
D4/LC5 (µg/l)	0.46 (drift)	2.32 (drainage)	0.84 (drainage)	0.46 (drift)	0.84 (drainage)
D2/LC6 (µg/l)	8.61 (drainage)	22.95 (drainage)	9.76 (drainage)	0.67 (drift)	2.94 (drainage)
D5/LC9 (µg/l)	0.49 (drift)	2.08 (drainage)	0.62 (drainage)	0.49 (drift)	0.50 (drift)
R1/LC8 (µg/l)	0.35 (drift)	2.07 (runoff)	2.40 (runoff)	0.59 (runoff)	0.98 (runoff)

6. Factors that influence ecosystem recovery

The ability of a species to recover from a pesticide impact will depend on its autecological and life history characteristics (sensitivity, exposure, recovery potential etc.), and its habitat (e.g. waterbody type, density). These factors are summarised below. Annex 6 provides a broader discussion, together with examples, of the recovery potential of selected taxa, waterbodies and landscape types.

6.1 Sensitivity and exposure

The sensitivity of a species to the lethal, sub-lethal and indirect effects of a pesticide are fundamental to recovery because they determine the base level of damage from which populations will recover. In addition, sub-lethal impacts often have direct implications for recovery rate, particularly where they affect a species' reproduction potential, either positively or negatively.

The majority of plants and invertebrates occupy a range of habitat niches during their lifetime. A species' spatial and temporal life history characteristics will, therefore, determine the extent to which it is exposed to agrochemicals. For example, insect species with a terrestrial phase are likely to be at lower risk of severe population damage where individuals emerge over a long period. This is because at any one time, the population will be partitioned into aquatic and terrestrial habitats at differing risk from impacts. It is important, however, not to assume that species that have emerged from the aquatic environment can be ignored for risk assessment purposes. In practice, the aerial phase may be highly vulnerable, particularly where species disperse directly across crop lands.

6.2 Recovery

6.2.1 Reproduction

A species' reproductive strategy determines the theoretical maximum rate at which populations can recover following an impact. Overall reproductive strategy is determined by the number of offspring produced per adult and the frequency of reproduction. Combining these data it can be generally assumed that:

- Species likely to recover more rapidly from impacts will be those with: (i) high production rates; (ii) short generation times; and (iii) flexible life history strategies. This includes both species that can respond rapidly through asexual reproduction (e.g. body division in flatworms) and multivoltine taxa with short life-cycles such as chironomid midges.
- Species likely to recover more slowly following a pesticide impact are those with: (i) low production rates; (ii) longer generation times, including a longer time to maturity; and (iii) more rigid life history strategies. This includes groups such as dragonflies and bivalve snails, some of which take two or more years to reach reproductive age.

6.2.2 Immigration

Immigration to an impacted site can occur from less affected parts of the same waterbody and/or by immigration from other waterbodies. Plants and invertebrates have different strategies to enable them to achieve these different forms of re-colonisation.

Between waterbody immigration: Between waterbody dispersal is generally achieved either by active flight (for adult aquatic insects), or by passive transport using animal vectors. More uncommonly, wind, and on floodplains, water, can be an important means of transporting plant and animal propagules between waterbodies. Biota can be broadly ranked in terms of their potential for colonising new sites. In general, winged insects with a strong flight tendency (e.g. bugs, beetles) have the greatest colonisation potential. Animals with no flight potential (e.g. leeches, crustaceans, snails) are slower colonists because they rely on less predictable means of passive dispersal. Measurements of average migration distance and probability of colonisation are exceptionally rare for most freshwater species. Direct observation and data from genetic studies are available for a few taxa, mainly active colonisers. The best information for passive taxa generally comes from studies of the species colonising new sites.

Within waterbody immigration. Plants and invertebrates use the full range of methods employed for *between* waterbody colonisation as methods for *within* waterbody dispersal. However, they also use a range of additional dispersal modes. In running waters, downstream drift is a highly effective method of recolonising impacted sections, although other colonisation modes are also used. In still waters, invertebrates generally disperse by active crawling or swimming, and plants by wind drift across the water surface. Plants in still and running waters also have the advantage of a semi-persistent seed bank to enable rapid re-growth. Many taxa that are poor at colonising other waterbodies can recolonise areas of their natal waterbody relatively rapidly. This means that within-waterbody colonisation will usually be both more rapid and more predictable than between waterbody colonisation.

6.3 The influence of waterbody type on recolonisation

The physico-chemical characteristics of a waterbody will significantly influence recolonisation. Factors such as waterbody size and connectivity will, for example, affect the availability of refuge areas within a waterbody. Thus small habitats such as ponds with few internal refuges may be at greater risk of pesticide impacts than lakes or running waters, because ponds may need to re-stock using the relatively unpredictable process of immigration from other waterbodies. Where this is the case, environmental variables such as waterbody density in the surrounds will have a significant influence on both the taxa that recolonise and the rapidity with which they do so. Finally, other factors such as habitat quality and disturbance history have the potential to play a role in recovery. Thus waterbodies that are regularly disturbed by anthropogenic or natural events, frequently have characteristic assemblages dominated by taxa that are naturally effective colonisers. This may give them a better recovery potential than less disturbed or more pristine habitats dominated by species with a broader range of recovery strategies.

7. The potential for the use of ecological information in probabilistic risk assessments

There is widespread recognition that variabilities, inaccuracies and approximations (uncertainties) are inherent in risk assessment procedures and that results from current deterministic (single outcome) methods are estimates for which the associated error is unknown. More effective regulation would result if these errors were quantified and outputs from modelling at higher tiers moved from a single value to a range of outcomes defined by a probability distribution. Most of the measurements presented in the database are technically suitable for inclusion in probabilistic risk assessment. Categorical data are summarised as proportion of sites falling into specific categories. Probability distribution functions can be defined by specifying the number of samples to be taken from each category and sampling from a uniform distribution within the category. All numeric data are provided as mean, median, standard deviation and range. Thus pdf's can be specified for normal or log-normal distributions or a triangular distribution may be specified based on the median and range where there is no evidence for a more specific distribution. There are two main constraints on the use of data in PRA. First, some of the datasets from which summary statistics are derived are rather small, particularly for ditches (e.g. see Table 5); number of samples is always provided in the database to inform decisions on utility. Secondly, the database contains derived data based on databases which have generally not been collected or quality assured by the contractors; thus there is always potential for errors or nonsensical values.

To demonstrate the potential use of information from the database in PRA, an example calculation was undertaken for estimation of aquatic exposure arising from spray drift. Predicted environmental concentrations (PEC's) from spray drift are usually calculated for ditches because they have limited potential for dilution of residues and calculations are based on several worst-case assumptions. The calculation was undertaken for landscape 2 as this is one the landscapes with ample data for ditches. Probabilistic drift calculations were made on the basis of summary statistics (mean, median, range and standard deviation) for: (i) distance from crop to top of bank; (ii) slope and height of bank; (iii) width of water column (surface); (iv) depth of water column; (v) presence and height of intervening vegetation between crop and water. The drift percentile entering the ditch was sampled from the full distribution of values using the mean and standard deviation values presented by Ganzelmeier *et al*. (1995). Monte Carlo sampling was undertaken using the Crystal Ball software (Decisioneering 2000) with a total of 60,000 runs per simulation.

Results are provided in Figure 7. For the use pattern under investigation, the preliminary EU exposure assessment edict would generate a predicted environmental concentration (PEC) of approximately 8.9 µg/l for an unrestricted application and 1.8 µg/l assuming a 5 m no-spray buffer restriction. Assuming no restrictions on the application, the first-step PEC from spray drift lay on the 99.7th percentile of the distribution curves for river floodplains and fenland landscapes, respectively. With a 5 m no-spray buffer imposed, the basic regulatory value equated to the 98.6th percentile of the distributions. Results thus confirm that conditions equivalent to the regulatory scenario can be found in agricultural landscapes, but that occurrence is rare.

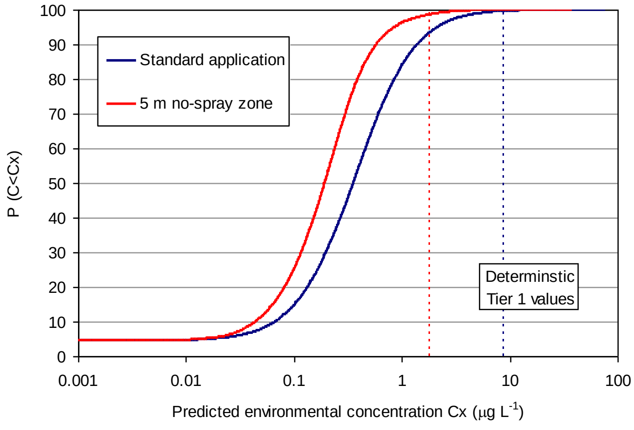

Figure 7. Probability distribution curves for PEC's in ditches in Landscape Class 2

8. The potential for use of spatially explicit tools such as GIS

Risk assessment for pesticides adopts a tiered approach. Generalised, protective scenarios are used at the lower tiers, whereas the assessment becomes more and more site-specific at higher tiers. There is currently considerable interest in the use of GIS and associated data collection methods to support risk assessment at the higher tiers and to facilitate post-regulatory management of risk. GIS may be used to interrogate and display inter-related data (e.g. for scenario definition or to derive inputs for models) or may be linked to models to express the spatial variation of exposure and/or risk. At present, the potential to use data collated by the project within GIS is rather restricted. The only spatially-explicit information generated by the project comprises the digital map of landscape classes. The map is static within the graphical user interface but can be made available as an independent data layer which could be incorporated into a GIS with Crown permission. The SEISMIC system has been developed by Cranfield University on behalf of DEFRA to hold spatial data layers and associated data relating to soils, aquifer boundaries, weather and cropping. The system allows the development of scenarios for pesticide exposure assessment and provides the environmental parameters required to run pesticide fate models. A research requirement has been put forward to DEFRA to incorporate the landscape map into SEISMIC in order to allow its interaction with the other spatial datasets held in the system.

All remaining output from the project is derived (aggregated) data that are not spatially referenced. Nevertheless, many of the raw data underpinning the aggregation are either held by the project consortium or publicly accessible and have been collated into one place for the first time. The site of each measurement is recorded, providing a background resource of information which could be implemented into a GIS to support risk assessment for pesticides or other activities. Some effort would be required to manipulate the data prior to use in GIS and there may also be restrictions on the use of some of the raw datasets.

9. Regulatory implications for pesticide risk assessment and management in the UK

9.1 The existing regulatory framework: current state and likely future trends

Standard exposure scenarios have been developed by the European Commission-sponsored FOCUS Surface Water Group in order to assist in the definition of 'one safe use' for Plant Protection Products prior to inclusion of active substances in Annex I of Directive 91/414/EEC. The remit of the FOCUS Surface Water Group was to produce a limited number of 'realistic worst-case' scenarios (maximum of 10), (see Annex 2 for scenarios). Collectively, these scenarios are intended to be representative of agricultural conditions throughout the Europe Union, but are not intended to be specifically representative of Member States. If these scenarios are to be used to inform regulatory risk assessment in the UK, then it is necessary to know both the range of relevant environmental characteristics within UK agriculture and to what extent these characteristics are included in the existing FOCUS scenarios.

The ecological data gathered in this project are likely to have particular relevance to the design and subsequent interpretation of higher–tier risk assessment strategies as discussed in the HARAP workshop (Campbell *et al.* 1999). Specifically, understanding the relevance of additional species to a range of UK waters may be critical to help reduce uncertainty in the risk assessment process. Application of a probabilistic approach to assess the risks of pesticides will require an understanding of the spatial and temporal nature of potentially exposed systems. The alignment of information on the spatial distribution of different types of water body in landscape classes, with cropping data and the use pattern of the product (GAP), will facilitate estimation of more realistic exposure scenarios for a particular pesticide (see also Section 7).

9.2 The realism of PEC calculations

Five of the thirteen landscape classes defined in this project are generally represented by existing FOCUS Surface Water bodies:

- LC3 (Sandlands) is similar to FOCUS drainage scenario D3 (Vredepeel)
- LC 5 (Oligotrophic till landscapes) is similar to FOCUS drainage scenario D4 (Skousbo)
- LC 6 (Pre-Quatenary clay landscapes) is similar to FOCUS drainage scenario D2 (Brimstone)
- LC 8 (Pre-Quatenary loam landscapes) is similar to FOCUS runoff scenario R1 (Weiherbach)
- LC 9 (Mixed hard fissured rock and clay landscapes) is similar to FOCUS drainage scenario D5 (La Jailliere).

FOCUS scenario D4 is generally representative of large areas of UK agriculture. However, this is partly a function of the coarse resolution in the European FOCUS scenarios that were primarily identified on the basis of very broad pedoclimate characterisation. The incorporation of water body characteristics such as distribution in the landscape, moving or static, hydrology and boundary morphology, vegetation barriers etc. in the modelling of PECs, should provide more realistic estimates of exposure.

9.3 Development of probabilistic approaches

Within HARAP it is suggested that data gathered on additional species can be employed within a probabilistic risk assessment. This technique allows toxicity data to be fitted to a statistical model to represent the distribution of sensitivities that would be expected in nature. SSD curves have been demonstrated to provide realistic estimates of the sensitivity of communities to exposure within mesocosms and thus provide a valuable tool in determining the likelihood of observing adverse effects within natural systems. The database for the current project shows that while some generalist taxa may be found in a wide range of waterbody types, other taxonomic groups are restricted to one or two water body types. Currently little is known about the specific sensitivity of many taxa found in these different habitat types, although ongoing research is addressing this issue (DEFRA project PS2304). Within the context of higher tier risk assessments, consideration should be given to the taxonomic composition of potentially exposed water bodies, and tests conducted on relevant taxa.

9.4 Regulatory implications of the study

Higher tier laboratory and field studies aim to reduce the uncertainty associated with extrapolating the observed responses for a few standard test species to the responses of communities exposed in natural systems. The selection of taxa for use in higher tier studies should, therefore, consider both the mode of action of a chemical, and the habitat and behaviour of the proposed test organism when determining the likelihood of, and response to, exposure ensuring that results are representative of sensitive taxa within a community.

Mesocosm studies expose whole assemblages to a chemical, allowing both direct and indirect effects of the chemical to be determined. Published research has shown variation in the extent to which mesocosm assemblages are representative of natural freshwater communities (e.g. Dyer & Belanger 1999, Williams *et al.* 2002). This project provides new information on the composition of macroinvertebrate and macrophyte assemblages found in ponds, ditches and streams in agricultural landscapes within the UK. It also provides an assessment of assemblage similarity both within and between water body types and landscape classes. This information can be used to determine the extent to which higher tier study designs do represent the distribution and sensitivity of potentially exposed communities in the wider environment reducing the uncertainty associated with extrapolating from the tested taxa to natural assemblages.

10. Discussion, implications and limitations

10.1 Map of landscapes and relational database

The definition of landscape classes developed for the project is a pragmatic one designed to capture broad differences in types, properties and abundance of waterbodies, potential for exposure from pesticides (i.e. agricultural land use) and routes of movement of water (and thus potentially pesticide) from agricultural fields to water. Analyses demonstrate that differences in these parameters are encapsulated in the landscape classification. The definitions are partially subjective. **Implication: the landscape classification is fit for purpose, but is specific to the separation of aquatic habitats likely to have different morphological, chemical and biological properties.**

The map of landscape classes was delineated on the basis of soil parent material. Although distributions on soil parent material correlated visually with those for hydrogeology, topography and land use, datasets specific to these latter parameters were not used in generating the map. Original linework for the map is at a scale of 1:250,000. The resulting map was rather fragmented and some smoothing was undertaken by (i) blending smaller polygons up to 2 km in diameter into larger polygons surrounding them; and (ii) removing long, thin polygons up to 500 m across. **Implication: the origin and scale of the map should be considered when using it to support data analysis and/or risk assessment.**

The project has collated a large number of databases and subjected them to further analysis and statistical summary. Only a limited check of the quality of the original data has been possible and only obvious mistakes are likely to have been picked up. In summary statistics, the range will be particularly sensitive to errors in the original data; the mean and standard deviation will be less sensitive; and the median is likely to be the most robust statistic. **Implication: end-users should undertake plausibility checks when using the data, particularly when working with the range of a given parameter.**

10.2 Biological communities of agricultural landscapes

<u>Invertebrates</u>

Evaluation of the species-richness of waterbody types in agricultural landscapes shows that all waterbodies contribute to alpha and gamma diversity; ponds are surprisingly rich; ditches are especially important in LC 2 and possibly other land classes for which there are currently no data. In terms of uncommon species, ponds made the greatest contribution. Targeted surveys (Coleshill, Oxfordshire and Whitchurch, Shropshire) broadly confirm these results. **Implication: both large and small waterbodies make an important contribution to biodiversity; ditches, although entirely artificial habitats, can sometimes be exceptionally biodiverse.**

Rivers, ponds and ditches showed significant differences in species richness between landscape classes. In rivers this was mainly due to the richness of LC1; in ponds mainly due to a pH gradient differentiating the upland from the lowland landscapes; in ditches most differences were due to the richness of LC2. The latter reflected the fact that ditches in this landscape were often permanent and had historic continuity with ancient wetlands created following the deglaciation of the British Isles. Streams showed no difference in species richness between landscapes. **Implication: in lowland landscapes species richness of ponds and streams differs little between landscape type; rivers in LC1 are significantly richer than other landscape classes but this may be an artefact of size; LC2 ditches are particularly rich.**

Rare species occurred in similar proportions in all landscapes in streams and ponds. Limited evidence suggests that rivers also supported similar proportions of rare species irrespective of landscape type. For ditches LC2 supported more rare species than other landscape classes. This is not unexpected since fen ditches are well known for their tendency to support rare species. **Implication: rare species are equally likely to be found in streams, ponds and rivers, irrespective of landscape class. Fen and warpland ditches often have an unusually high proportion of uncommon species and should be given special attention.**

The composition of the invertebrate fauna varies markedly between waterbody types. Ponds are dominated by water beetles and water bugs; in streams and rivers caddis flies, stoneflies and mayflies represent a larger proportion of the fauna. This has significant implications for the proportions of the fauna made up by pesticide sensitive species in different landscapes, specifically:

- Rivers have most pesticide sensitive species, followed by streams; ditches and ponds have fewer sensitive species
- The proportion of pesticide sensitive taxa in different landscape classes varies in streams (upland sites have a larger proportion than lowland sites). Proportions of sensitive taxa remained constant in ponds in different landscape classes, and possibly also in rivers and ditches though too few data were available to be certain of this.

Note that the areas of the landscape which naturally have fewer pesticide sensitive species (e.g. lowland streams) also have greatest exposure to biocides. This makes it difficult to determine whether the observed proportions of sensitive taxa are the result of natural habitat factors or to the existing levels of exposure to pesticides. **Implication: flowing waters have more sensitive species than still waters; in more acid sites pesticide sensitive taxa are likely to make up a larger proportion of the fauna.**

Macrophytes

National macrophyte data could not be used for between waterbody comparisons because they were collected using incompatible survey methods (e.g. 20 m lengths from ditches compared to 1000 m lengths for rivers). **Implication: it is not currently possible to use national plant datasets to make reliable comparisons of richness *between* waterbody types.**

Regional datasets collected for the project using compatible methods suggest that, for macrophytes, rivers were the richest waterbody type at site level but that regionally, ponds are likely to be an equally important biodiversity resource. All waterbody types contributed to regional floral diversity however. **Implication: maintenance of macrophyte biodiversity requires protection of all waterbody types.**

Between landscape differences in macrophyte species richness could be detected in all waterbody types; in part this was probably due to the larger macrophyte sample sizes but may also have reflected the fact that, for macrophytes, the difference between the richest and poorest landscapes was consistently greater than for macroinvertebrates. In addition, LCs 1 and 2 were consistently amongst the richest for macrophytes. Between-landscape differences in Species Rarity Index values were also significant and, with the exception of lakes, were generally highest in LCs 1 and 2. **Implication: particular attention should be given to the protection of macrophyte assemblages in LCs 1 and 2.**

10.3 Risk assessment scenarios and higher tier assessment

The project presents a large amount of information that can be used to derive risk assessment scenarios either for specific compounds or generically. This includes data on cropping, potential pesticide inputs, soil types, routes of exposure, morphology and chemistry of waterbodies, assemblage characteristics and presence of sensitive species and internal and external recovery potential. It is currently impossible to include more than two to three interacting factors when weighing the relative vulnerability of different scenarios. Potential steps in defining scenarios for risk assessment are: (1) generate a problem formulation setting out the issue to be addressed; (2) list the factors likely to impact on the risk, rank the factors in order of relative importance and note any interactions between them; (3) use the database to rank the relative vulnerability of different waterbodies and/or landscape classes for the few most important factors; (4) use information from 2 and 3 to justify the selection of one or more assessment scenarios. **Implication: definition of risk assessment scenarios will remain somewhat subjective, but results from the project can be used to increase the objectivity and the confidence associated with scenario selection. Overall vulnerability depends on many interacting influences and will vary according to the compound and use situation to be assessed.**

Output in the database is generally suitable for inclusion in probabilistic approaches to risk assessment, although summary statistics are dependent on the quality of the original data. Basic sensitivity analyses could be used to decide whether to use summary statistics for a particular parameter or to initiate measurement, reference to original databases or further analysis. The aggregated data are not suitable for use in GIS directly, but output could be used to target data collection for GIS applications. **Implication: project results are generally supportive of higher tier approaches to risk assessment. Where data cannot be used directly, they should help to target the collection of further data.**

10.4 Recovery

A review of the factors likely to influence the recovery of species and ecosystems after a pesticide impact indicate that recovery processes operate at a range of spatial and temporal scales.

At a macro-scale, waterbody type will significantly influence whether recolonisation occurs from within the same waterbody or by immigration from elsewhere. Of the two, internal colonisation is more likely to promote rapid recolonisation of a pre-impact assemblage than immigration. This is because: (i) species have fewer means of moving between sites than within them; and (ii) for between waterbody movements there are considerable differences in species mobility so, although common mobile species may colonise new sites rapidly, species that are uncommon or have low mobility may not recolonise for a considerable period, if at all.

In running waters, the high connectivity and availability of upstream refuges means that, regardless of landscape type, the majority of these waterbodies are likely to recover relatively rapidly from a moderate pesticide impact: usually in the order of 3 - 18 months. On this time scale, regular (e.g. seasonal) pesticide inputs would have the potential to permanently depress assemblage composition. Very occasional impacts would, however, be expected to have a relatively limited affect. Exceptions to this, where a slower return to the pre-impact channel biota would be predicted, are: (i) headwater areas with no upstream refuges; (ii) channel lengths of very high quality or of variable type, where species distributions may be highly localised; and (iii) all running waters subject to extensive pollution events (e.g. a major chemical spill or inputs over a considerable length) where a high proportion of pre-impact species are absent from upstream sources. In these cases, recovery will depend on both the between-waterbody mobility of these species and their density in the surrounds. This may lead to some differences in colonisation rate between landscape classes. Thus hardrock (LC10) and footslope loam (LC12) areas that are dominated by actively flying taxa, may colonise more rapidly than LC2 (fens and warplands) or LC7 (chalk and limestone) which have a higher proportion of passively colonising taxa.

For standing waters, larger waterbodies (e.g. lakes) are likely to have a relatively high degree of connectivity. Thus, most lakes subject to moderate pesticide impacts will retain unaffected areas that can provide an internal refuge source. The (limited) available evidence suggests that lateral movement within still waters can be relatively rapid, so recolonisation of lakes *might* be achieved over rather similar timescales to streams and rivers. Small standing waters (e.g. ponds, seasonal pools) will generally have a lower potential for internal buffering because a higher proportion of the waterbody is likely to be affected by pesticide impacts. In such circumstances, recolonisation will at least partly occur from external sources. For active colonisers (i.e. those that are winged and fly readily) the probability of site colonisation is high and potentially predictable for most species. For taxa spread by passive dispersal, however, colonisation is more open to chance events (e.g animal and bird movements), and less likely to result in a return to the pre-impact assemblage. In ponds, species composition is similar across the different landscape types, so this is not likely to be an important factor in pond recovery. **Implications:**

- **Recovery from moderate pollution events to a pre-impact assemblage is likely to take longest for habitats such as (i) small waterbodies (e.g. ponds, temporary pools) (ii) headwaters, and (iii) relatively isolated or upper catchment ditch complexes**
- **For ponds, in particular, initial colonisation of mobile taxa may be rapid, but stochastic processes may mean that the returning assemblage is different to the pre-impact community.**
- **Significant pollution events would, in addition, have the potential to disrupt larger waterbodies e.g. lakes, rivers. Since these habitats are often relatively sparse in the landscape, their recolonisation period could be protracted.**
- **All waterbodies are at risk from prolonged press or regular pulse pollutant impacts, where there is insufficient time for recovery processes to operate effectively.**

At a meso-scale, habitat factors will affect recolonisation potential. Thus, within waterbodies the patchiness of pre-impact species distributions will influence the rate at which recolonisation can occur by internal migration. Habitat quality and within waterbody feedback effects also occur at this scale. As the scale considered becomes smaller still, it becomes more difficult to make generalisations about the rate of recovery. At a micro-scale, there are an enormous number of variables that affect the recovery potential of species. This includes the natural complexity of population and life-history dynamics including both (i) species-dependent variables (e.g. production, habitat, mobility), and (ii) environmental variables (levels of predation, competition and disease, climate differences etc.). These variations are compounded by the chronic and indirect effects of the pesticide itself. For many of these factors, and for many species, few data are available. **Implication: given a sufficient number of scenarios it may be possible to make generalisations about the likely recovery of different waterbody types. However, the considerable number of meso and micro-scale variables influencing population recover suggest that attempts to make detailed and reliable predictions of recovery rates of specific populations of plants and animals are likely to prove challenging.**

10.5 Implications of the study for design of laboratory and experimental field studies

There is general acceptance that the pesticide risk assessment process should focus on species likely to be most sensitive to pesticides. This study shows that taxa which are generally more sensitive that then the standard Tier 1 invertebrate test species (*Daphnia*), are widely represented in all waterbody and landscape types. These taxa (stoneflies, mayflies, amphipod crustaceans), make up a moderate proportion of the fauna of ponds and ditches (5%-10%), but they are significantly represented in streams and rivers (20%-40%). In addition, the project's regional surveys suggest that the proportion of sensitive *individuals* in these groups may be considerably higher for all waterbody types (c. 40% - 60%). **Implication: stoneflies, mayflies and amphipod crustaceans should be adequately represented within pesticide risk assessment.**

Density calculations for the waterbody types confirm that ditches and ponds are the generally the most abundant waterbody types in agricultural landscapes. Streams and rivers do, however, comprise a significant component of some landscape classes (e.g. LCs 1, 5, 9). An implication from this is that a running water habitat element would be appropriate for higher tier risk assessment. In practice, this may not prove practicable in all cases since functional stream mesocosms required for outdoor multispecies tests can be difficult to create and maintain. It does, however, reinforce the need to ensure that there is adequate representation of sensitive taxa in static systems. This may be achieved by ensuring that, within sensitive groups, facultative species that can live in both running and still waters are represented in static mesososms **Implication: include running water mesocosms in higher tier risk assessments, or at least ensure that ensure that within static systems, sensitive species are adequately represented.**

11. The extent to which the objectives have been met

The major project objectives were fully achieved. Limitations in data availability did, however, mean that it was not possible to collate and analyse biotic, morphological and physico-chemical data for all waterbody types and landscape classes. These exceptions are documented in relevant sections of the report, and discussed in the recommendations for further research.

12. Recommendations for future research

12.1 The quality of information describing the aquatic ecosystems of the agricultural landscape

The data collated for the current project were sufficient to give a first approximation of the biotic composition of waterbodies in a range of agricultural landscape types. There remain, however, a number of critical gaps where additional data are necessary for an adequate understanding of freshwater assemblages in agricultural areas. Specifically:

(i) *Ditch data* are exceptionally scarce for most agricultural landscape types. This makes it difficult to develop risk assessment scenarios for the waterbody type that is found most frequently in agricultural areas. Ditches are small in size and their limited dilution potential means that they are often the main waterbody considered in risk assessment for pesticides. There is a need for systematically collected ditch data, including seasonal ditches, across a range of landscape types. The availability of *lake invertebrate* *data* is currently even more limited. Lakes are, however, of little relevance to risk assessment for pesticides under 91/414. In addition, at least some lake invertebrate data are likely to be gathered in future years to fulfil Water Framework Directive targets.

(ii) *Systematically collected plant data*. Current national plant datasets have been gathered using different methodologies. To enable direct richness comparisons to be made between waterbody types, it is necessary to extend the regional surveys used in the current project to cover a greater proportion of agricultural landscape classes.

(iii) *Minimally impaired site data*. The datasets used in the current project were predominantly from agricultural landscapes. This gives an indication of the *status quo*, but the landscape patterns described here are likely to be modified by the effects of diffuse agricultural pollution (including pesticides, fertilisers, sediments) and by waterbody isolation. A more reliable indication of the natural structure of aquatic assemblages, particularly in the absence of pesticides, can be derived from aquatic habitats in semi-natural landscapes protected from agriculture, so called 'minimally impaired' or reference sites. For example, examination of pond assemblages, for which both impacted and minimally impacted data are available, suggests that pond richness and rarity values are generally greater, and also more variable between landscape types, in areas free from agricultural pollutants. There is a need, therefore, to assess the difference between assemblages of impaired and minimally impaired landscapes. Data on such sites are available for pond plant and macroinvertebrate assemblages; they have also been collected from rivers and streams but this dataset (which is held by NERC and CEH) was not made available for the current project. As noted above, no systematically collected ditch data are available.

(iv) More information on *test species*. For all waterbody types, data are needed for taxa that are an important component of pesticide risk assessments, particularly zooplankton, Diptera and fish. At present data are available for some waterbody types for some of these groups (e.g. fish in rivers and streams, zooplankton in ponds). It is recommended that a review of the availability and accessibility of these datasets (e.g. are datasets digitised?) is undertaken.

(v) *Regional studies*. In the present project relatively small-scale regional studies have provided a valuable complement to the review of major existing national datasets. Regional studies have the advantage that they can be: (a) specifically designed with the requirements of pesticide risk assessment in mind (e.g. incorporating ditches) and (b) provide data which are directly comparable between different waterbody types. Further development of the regional datasets, targeted on the landscape classes identified in the present study, could provide a highly effective method of refining understanding of the aquatic habitats of the agricultural landscape, rather than relying on national datasets which are typically collected for general environmental monitoring purposes.

12.2 Research to forward the development of PRA, and development of more realistic test scenarios

(i) The SEISMIC system has been developed by Cranfield University on behalf of DEFRA to hold spatial data layers and associated data relating to soils, aquifer boundaries, weather and cropping. The system allows the development of scenarios for pesticide exposure assessment and provides the environmental parameters required to run pesticide fate models. A research requirement has been put forward to DEFRA to incorporate the landscape map into SEISMIC in order to allow its interaction with the other spatial datasets held in the system.

(ii) Several research requirements arising out of the current project are being addressed as part of DEFRA project PS2304 (WEBFRAM2: predicting the effects of pesticides on non-target aquatic organisms). These include: the need for life history information for species sensitive to pesticides; the need for information on the proximity of ditches to agricultural land and the need to generate reference images of species assemblages for aquatic habitats in the UK. These research requirements are not discussed further here.

(iii) Additional information is required about the macro-scale factors affecting the recovery of sites, particularly the recovery of still water sites though internal and external colonisation. Specifically: (i) a review of available grey data held by agrochemical companies on mesocosm recolonisation, and (ii) full scale experimental manipulation of ponds and ditches to investigate recovery potential in field situations.

(iv) Ultimately, field studies need to be combined with modelling of meta-population level impacts of biocides to obtain general theoretical understanding of the point at which multiple impacts in a catchment context begin to significantly impact on assemblage composition and population sizes.

12.3 Research to improve understanding of the field impacts of pesticides

The demonstration of probabilistic risk assessment (PRA) methods in the present project suggests that, under present regulatory guidance, exposure to pesticides in the field is likely to be at a low level. There are, however, a number of studies that have detected pesticide related effects on biota at very low concentrations. This indicates the importance of further work in this area and, particularly, for the general protectiveness of existing regulatory measures to be confirmed in the field. There are currently very few field data that can be used either to corroborate laboratory and mesocosm results or to establish the *relative* importance of pesticide and other diffuse pollution impacts in aquatic habitats influenced by agricultural ecosystems.

13. Publications arising out of the project

Biggs, J., Whitfield, A.M., Williams, P.J., Nicolet, P., Arnold, D., Bellamy, P., Brown, C., Hollis, J., Lyons, H., Maund S J., Pepper, T. and Turner, N. (in prep). Characterisation of aquatic habitats in the British agricultural landscape: implications for pesticide risk assessment.

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