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Morphological and physico-chemical properties of British aquatic habitats potentially exposed to pesticides

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Abstract

Approaches to describe the exposure of non-target aquatic organisms to agricultural pesticides can be limited by insufficient knowledge of the environmental conditions where the compounds are used. This study analysed information from national and regional datasets gathered in the UK describing the morphological and physico-chemical properties of rivers, streams, ponds and ditches. An aggregation approach was adopted whereby the landscape was divided into 12 hydrogeological classes for agricultural areas and a 13th class that comprised non-agricultural land. The data describe major differences in the abundance, dimensions and chemistry of water bodies in the different landscapes. There is almost an order of magnitude difference in the total input of pesticide per unit area between the different landscapes. Ditches are shown to be most proximate to arable land, streams and rivers intermediate and ponds the least proximate. Results of the study have implications for the development of standard scenarios for use in protective screening steps within the risk assessment. Data can be used to produce more realistic estimates of the exposure to pesticides and to examine how that exposure varies across the landscape.

1 Introduction

The use of pesticides on agricultural land may result in contamination of adjacent surface waters and thus pose a potential risk to a range of aquatic organisms. The predominant routes of entry arising from diffuse applications of pesticide include spray drift, surface runoff and leaching to field drains. Systems of regulation for pesticides prior to approval for use include a demonstration that unacceptable risk will not be posed to the aquatic environment. An assessment is undertaken which often uses models to predict concentrations of a particular compound in different water bodies and combines this with ecotoxicological information to derive a measure of risk.

The environmental fate of pesticides is influenced by a large number of factors including the inherent properties of the chemical, the timing and pattern of use of the product, the behaviour of the farmer and the performance of farm equipment. Fate is also determined by a range of environmental parameters related to soils, hydrogeology, topography, crop physiology, weather and irrigation. Given the complexity of these interacting factors, the most common approach to modelling fate has been to derive a limited number of environmental scenarios which are then used as the basis for predicting the behaviour of individual pesticides and comparing between different compounds. A standardised modelling approach for the calculation of exposure of surface waters in Europe has recently been established by FOCUS (2003). A modelling tool was developed which combines mechanistic models with 10 standard environmental scenarios selected to represent the European agricultural area. The approach is designed to produce a generalised expression of exposure whose conservatism derives from worst-case assumptions built into the calculation. For example, the FOCUS tool assumes that water bodies adjacent to agricultural land are either ditches, streams or ponds and that each of these categories can be described with a single set of dimensions throughout Europe. Thus the FOCUS stream is 1 m in width, 0.3 to 0.5 m in depth, 100 m in length and has an average residence time of 0.1 day; there is a 5 cm layer of sediment at the base which has 5% organic carbon; the stream is situated 1 m away from a 1 ha treated field; and it receives water and pesticide from a 100 ha upstream catchment which has the same characteristics as the treated field, but only 20 ha are treated with pesticide (FOCUS, 2003).

Predictive modelling based on scenarios has several advantages in that it is relatively simple and quick to undertake, there is a high degree of standardisation between chemicals and, provided scenarios are appropriately defined, the approach can provide a conservative assessment of exposure and thus a high level of environmental protection. Unfortunately, as there are many interacting factors, it is not usually possible to quantify the likely level of conservatism. For example, it is not simple to quantify the likely impact on exposure and ecological effects of a stream which is shallower (i.e. less immediate dilution potential) but faster running (i.e. greater advective losses from the system) than the standard scenario.

In practice, there is a sparcity of readily-available data on the properties of aquatic habitats. This creates two problems. First, the definition of assessment scenarios becomes a rather subjective process which is not underpinned by detailed analysis of the true range of environmental conditions. There is a tendency to introduce overly protective assumptions in the absence of detailed information, but it is very difficult to assess the realism of the scenario and the degree of protection afforded. Secondly, there is no solid basis on which to proceed to more realistic assessments of exposure.

A number of workers have reported predictions of the spatial and temporal variation in pesticide concentrations at the watershed level (Bach et al., 2001; Cryer et al., 2001; Dabrowski et al., 2002; Verro et al., 2002). The approaches combine mathematical models with a geographical information system and address the variation in exposure arising from differences in soil type, topography, land use and climate. Variability in the properties of the receiving water bodies cannot be considerd at this scale. There have also been significant advances in using remote or aerial imaging technologies to investigate the way that agriculture interacts with the landscape (Hendley et al., 2001; Padovani et al., 2004). These generally consider a specific location and have tended to concentrate on determining proximity and density of arable cropping adjacent to water bodies. Information on a broader range of properties of water bodies is collected in surveys by groups from government, nature conservation agencies and the research community. However, the surveys are often only reported within the grey literature and the resulting databases are disparate in terms of format, parameters held and physical location.

In this paper, data from major national freshwater datasets in Britain are used to present an initial characterisation of the physico-chemical and morphological properties of aquatic habitats in the agricultural landscape. A hydrogeological classification of landscape is used to group water bodies into broadly similar types and to allow comparison between landscapes. Biological data from these habitats are presented in an accompanying paper (Biggs et al., 200x).

2 Methods

2.1 Derivation of landscape classes

Working definitions were developed for the four waterbody types included within the analysis: ditches, ponds, streams and rivers (Table 1). 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 for the study; and (iii) criteria that could be derived or calculated from Ordnance Survey maps.

Landscape classes were defined 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 a classification of soil types according to their hydrological response (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.

2.2 Data collection and processing

2.2.1 Abundance of water bodies

The spatial abundance 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 (Firbank et al., 2003). The 569 squares (each 1 km^2) of the Countryside Survey were reclassified into the 13 landscape classes. The mean length of ditch and stream per km^2 and the mean density of ponds per km^2 was calculated for each class.

2.2.2 Morphological and physico-chemical properties of water bodies

Pre-existing datasets were accessed and merged where possible to describe the morphological and physico-chemical properties of the different waterbodies (Table 2). Datasets were filtered to exclude monitoring sites potentially impacted from urban or industrial situations. 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 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). Categorical data were summarised as proportion of sites falling into specific categories. All numeric data were described by mean, median, standard deviation and range.

Additional field data were collected to address a lack of adequate information describing ditches in agricultural landscapes. New ditch data were gathered within a 10 x 10 km area at four field study sites in contrasting agricultural landscapes. The sites and grid reference of the north-west corner of the experimental area were: Spalding, Lincolnshire (LC2; ****), Morpeth, Northumberland (LC4; ****), Whitchurch, Cheshire (LC5; ****), and Kington, *** (LC7; ****). From each area, physico-chemical field data were collected from 10 randomly located ditch sites including ****. Need grid references and further details of ditch survey work

2.2.2 Land use and potential for exposure to pesticides

A spatial dataset for agricultural land had previously been produced by combining Agricultural Census data for Great Britain for 1995 (http://datalib.ed.ac.uk/EUDL/agriculture/) with a remotely-sensed Land Cover Map of Great Britain (Fuller et al., 1994). This dataset was overlaid onto 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.

Field size – need methodology for this

The distance between a water body and the nearest cropped land receiving inputs of pesticides is a key determinand for potential exposure, especially for the more localised transport processes such as spray drift. Many of the datasets contained fields descriptors for land use adjacent to water bodies, although the actual measure varied greatly. The area extending 50 m from rivers and streams was characterised as to whether different land uses were ‘absent’, ‘present’ or ‘extensive’. The proportion of different land uses was measured for a 5 and 100 m radius around ponds. Ditches are the most intimately associated with agriculture and here the distance to the nearest arable field was available. The data were collated to give a crude comparison of the density of arable cultivation around water bodies in the different landscapes.

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 (Reference).

2.2.3 Database construction

A relational database was constructed as a repository for the processed (aggregated) data on the properties of the landscape classes. Data tables were imported 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 interrogation of the map of landscape classes. The database and GUI are Windows-based software designed for use with either Windows 2000 or Windows XP platforms. The database is available for free download at: ftp://ftp.silsoe.cranfield.ac.uk/public/aquatic/.

2.3 Statistical analysis

Statistical tests used for hypothesis testing were performed using the statistical software program Statistica, version 6.1 (Tulsa, OK). The exception was for χ^2 tests, which were calculated longhand using significance tables given by Kanji (1999). Differences between sample means were analysed using t-tests or 1-way ANOVA, with post-hoc Tukey HSD tests used to identify significantly different sample means. Where data were non-normal (determined by the Kolmogorov-Smirnov test for normality) or the variances of the sample data differed significantly (determined by F-tests), non-parametric methods were used to look for differences between sample medians. Kruskall-Wallis tests were used as an equivalent to one-way ANOVA, with post-hoc testing performed using Mann-Whitney U-tests. Correlation analysis between non-normal datasets was performed using Spearman's rank correlation. Differences between distributions were assessed using χ^2 tests. Statistical tests were considered significant at the 95% level ($p < 0.05$), and these probability levels are implied unless otherwise stated. As the distinction between streams and rivers is operational, these two types of waterbody were not differentiated for any of the chemical and morphological analyses.

2.3.1 Waterbody morphology

River and stream (combined) water width and depth measurements were analysed to assess whether morphological differences existed between landscape classes. As the datasets for each landscape class were strongly skewed towards zero (determined using Kolmogorov-Smirnov normality tests), non-parametric statistics were used. Where significance ($p < 0.05$) was found following Kruskall-Wallis tests, landscape classes were ordered according to width or depth rankings, and pair-wise comparisons made between landscape classes using Mann-Whitney U-tests.

As raw width and depth data were identified for each sample site, it was possible to perform correlation analysis between these data using Spearman's ranked correlation test. Where significant ($P < 0.05$) correlation between width and depth was demonstrated for a landscape class, determination of a cross-sectional shape parameter (depth / width × 100) for each landscape class could be justified. Large values for this parameter indicate a deep, narrow water body. Differences between medians for each landscape class were determined as for width and depth.

Pond surface area and average depth measurements were statistically analysed to determine whether morphological differences existed between landscape classes. Surface area and depth were correlated using Spearman's rank correlation, and medians of pond volume (surface area × depth) were analysed using a Kruskall-Wallis test.

Ditch width measurements were statistically analysed for differences between waterbodies. As the raw data were categorical, a χ^2 test was used to look for differences in category distribution between landscape classes.

2.3.2 Bed substrata of rivers and streams

Detailed information on stream/river-bed material was analysed to look for differences in distributions of bed material between landscape classes.

2.3.3 Waterbody chemistry

Water chemistry data for streams and rivers (combined), ponds and ditches were analysed for differences between landscape classes (within waterbody groups), and differences between waterbody types using one-way ANOVA tests. Comparisons between all waterbody types are presented for pH and conductivity.

3 Results

3.1 Identification of landscape classes potentially exposed to pesticides

A total of 12 agricultural landscape classes was identified for England, Scotland and Wales, with a thirteenth class comprising all non-agricultural land (including urban, forestry, non-maintained grassland and amenity uses). Table 3 summarises the properties of the classes and their spatial distribution is shown in Figure 1. The number of sampling sites for each waterbody in each landscape class is shown in Table 4. Physico-chemical data relating to the specific Scottish landscape classes (11 and 12) are limited and these classes are not considered further within this study. However, these classes are included within an accompanying analysis of the biology of freshwater habitats (Biggs et al., 200x). Many of the other landscape classes are also present in Scotland.

3.2 Abundance of water bodies

Relative distributions of rivers, streams, and non-road ditches (defined as average length of waterbody (m) per km^2) and the number of ponds per unit area are shown in Figure 2. The frequency distribution of rivers, streams and ditches was shown to be highly significant using a χ^2 test ($\chi^2_{\text{test}} = 7044 > \chi^2_{\text{critical (p = 0.05)}} = 31.5$; 18 degrees of freedom). The greatest contributor to the large χ^2 value was from Landscape 1 (floodplains) which explained 34.8% of the overall difference between observed and expected values. Landscape 1 is by definition the most dominated by rivers, whereas streams are more evenly distributed amongst the landscapes. Streams and rivers are least abundant in chalk and limestone areas. Ditches are the dominant feature in landscapes 2 and 6 (fenlands and clay areas) and they are also particularly numerous in non-agricultural areas, presumably in parts that are too wet to support agriculture. The number of ponds per unit area varied six-fold between landscape 5 (low base tills) and landscape 6 (clays). Average pond size varied between 0.07 and 0.37 ha, with the largest ponds in landscape 10 (hard rock).

Data from Countryside Surveys undertaken in 1990 and 2000 indicate that the area of streams, rivers, ditches and lakes did not change significantly during this period. 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).

3.3 Land-use and potential for exposure to pesticides

Agricultural land-use across the 11 landscape classes is shown in Table 5 alongside average field sizes for the different landscapes. A simplified characterisation of land-use is provided in Figure 3. Field size is closely related to land use. A correlation analysis indicated a positive correlation between field size and proportion of land under arable cultivation (Spearman's $R=0.915$; $p<0.05$).

Figure 4 shows the different measures for extent of arable cultivation around water bodies. Results for rivers and streams were almost identical, with only slightly denser arable land use around streams compared to rivers; these two datasets were thus combined. Across all landscapes, arable land use was 'present' or 'extensive' within 50 m of the river or stream for less than 50% of the sites surveyed. Arable cultivation tended to be either 'absent' or 'extensive' with few sites categorised at the intermediate level of arable land use 'present'. The land use around rivers and streams followed that in the broader landscape with the greatest amount of arable land in landscapes 2, 4 and 7 (cf. Figure 3). Other land uses such as orchards which receive inputs of pesticides accounted for less than 1% of land within 50 m of rivers and streams.

Arable fields were almost never located within 5 m of ponds, with the greatest exception being in landscape 4 (eutrophic tills) where surface-fed ponds within fields are a feature of the landscape. There was large variability in the amount of arable cultivation within 100 m of ponds, but on average this accounted for between 1 and 27% of the area for the different landscapes. Ditches tend to be intimately related with agricultural production. The average distance to an arable field was in the range 1.5 to 3.2 m, although the survey size was very limited.

Information on cropping patterns can be used with knowledge of any geographical or soil-related factors influencing use of a pesticide to estimate the variation in use within the different landscapes. Table 6 provides the average input of different types of pesticide to different landscapes as derived from pesticide usage statistics (reference). 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.

3.4 Waterbody characteristics

Analyses are presented for those chemical and morphological characteristics of waterbodies in the British agricultural landscape which have a bearing on the ecological risk from pesticides. The datasets are not equally extensive across all landscape classes for all parameters and for all water bodies. Therefore, data are presented only for landscape classes sufficiently represented (determined as sample sites > 10). For ditch data, where surveys were relatively constrained, coverage across the range of landscape classes is especially limited. Landscapes 11 and 12 (specific to Scotland) are not included in these results (therefore, a total of 11 landscape classes are considered). Note that other landscape classes are inclusive of Scotland (see Figure 1).

3.4.1 Waterbody morphology

<u>Streams and rivers</u>

Kolmogorov-Smirnov normality tests were performed on width and depth data for all landscape classes. For both datasets, all the data distributions differed significantly ($p < 0.01$) from the normal distribution. Kruskall-Wallis tests identified significant differences in median values between landscape classes for width data ($H_{(N = 8942)} = 1626$; $p < 0.001$) and for depth data ($H_{(N = 8911)} = 1253$; $p < 0.001$), enabling the classes to be ranked. Mann-Whitney U-tests were then used to differentiate individual landscape class medians. The relative rankings, median values and sample numbers are shown in Figures 5a and 5b. Landscapes 1 (floodplains), 2 (warplands / fenlands) and 3 (sandlands) include some of the widest and deepest streams and rivers, with landscapes 6 (clays) and 8 (loams) containing some of the narrowest and shallowest.

Spearman's rank correlation analysis was performed for width versus depth at each sample site, and in each landscape class. There was a highly significant ($p < 0.001$) correlation between width and depth for all landscape classes. It is thus reasonable to consider a river-bed shape parameter (S) defined as:

$$S = \text{depth (m)} / \text{width (m)} \times 100$$

These data were analysed as for width and depth. Kruskall-Wallis tests identified significant differences in median values between landscape classes for steepness data ($H_{(N = 8906, \text{ d.f.} = 10)} = 460$, $p < 0.001$), and Mann-Whitney U-tests were used to differentiate individual landscape class medians as before. The relative rankings, median values and sample numbers are shown in Figure 5c. Landscape 2 (warplands / fenlands) showed the greatest depth:width ratio, with landscapes 9 (rock & clay) and 5 (oligotrophic till) the shallowest. This analysis clearly separates landscape 2 from landscapes 1 and 3. The shape parameter is likely to be correlated with topography with larger values for S in the flatter landscapes such as that of the warplands and fenlands.

<u>Ponds</u>

Kolmogorov-Smirnov tests for normality showed pond volume data to be significantly different from the normal distribution ($p < 0.01$) for all landscape classes with the exception of landscape class 2 which had a small sample size ($n = 11$). A Kruskall-Wallis test showed no overall significant difference in pond volume between landscape classes ($H_{(N = 271, \text{ d.f.} = 9)} = 8.35$, $p < 0.001$). Mean pond volume varied between 461 and 3528 m^3 in sandland and hard rock landscapes, respectively.

Spearman's rank correlation was performed for surface area versus average depth data for each landscape class. There was no significant correlation between pond surface area and depth for any landscape class apart fron LC7 (chalk & limestone plateaux; $R = 0.58$, $p = 0.008$).

<u>Ditches</u>

The dataset for ditch width had a limited spread across the landscape classes, with only four landscape classes sufficiently represented in the data ($n > 9$). Figure 6 shows the distribution of width categories between these four landscape classes (raw data for ditch width were categorical). A χ^2 analysis was performed to determine whether differences in width distributions between landscape classes were significant. The four categories shown in Figure 9 were combined into two groups for the analysis: 0 to 3 m and over 3 m. The frequency distribution of ditch width between landscape classes was shown to be significant ($\chi^2_{test} = 17.7 > \chi^2_{critical\ (p\ =\ 0.05)} = 9.95$; 3 degrees of freedom). The biggest difference between observed and expected values (79.6% of overall difference) lies with landscape class 1 (floodplains), where more ditches fall within the narrower width category than for the other landscape classes.

3.4.3 Bed substrata of rivers and streams

The distribution of stream / river bed material types across landscape classes is shown in Figure 7. A χ^2 analysis showed that differences in bed material between landscape classes were highly significant ($\chi^2_{test} = 3069 > \chi^2_{critical\ (p\ =\ 0.05)} = 83$; 60 degrees of freedom). The biggest difference between observed and expected values (27% of overall difference) lies with landscape 2 (warplands / fenlands), where a high proportion (64%) of bed material is silt or mud.

3.4.4 Waterbody chemistry:

The mean values for pH and conductivity (showing associated error) for each waterbody are shown in Figures 8 and 9 for all landscape classes considered. Comparisons for streams and rivers were made for landscape classes 1, 2, 3, 10 and 13 only. One-way ANOVA on six key chemical determinants showed that there were no significant differences between landscape classes for conductivity, or for concentrations of suspended solids and Na. There were, however, significant difference for pH ($F = 14.9$; $n = 167$; $p < 0.001$), nitrite concentration ($F = 5.6$; $n = 164$; $p < 0.001$) and nitrate concentration ($F = 7.0$; $n = 167$; $p < 0.001$). Tukey's HSD post-hoc test for pair-wise comparisons between landscape classes for pH showed that the only significant individual difference between landscape classes was for landscape class 10 (hard rock), where mean pH was lower than for the other classes (pH = 7.1; see Figure 11).

Comparisons for ponds were made for pH, conductivity and Na concentration between all landscape classes except class 5 (insufficient data). One-way ANOVA showed that there were significant differences between landscape classes for pH ($F = 4.6$; $n = 240$; $p < 0.001$), conductivity ($F = 6.5$; $n = 253$; $p < 0.001$) and Na concentration ($F = 3.0$; $n = 241$; $p = 0.002$). However, there was no overall significant difference for pH when LC 13 (non-agricultural) was excluded from the analysis. The mean pH of 6.4 for LC13 was considerably lower than for the other classes. Tukey's HSD post-hoc test for pair-wise comparisons between landscape classes for conductivity showed that landscape class 13 (non-agricultural; lowest conductivity, 224 µs cm^{-1}) could be separated from classes 4, 3, 6 and 7 at the 95% significance level; landscape class 10 (hard rock; 252 µs cm^{-1}) could be separated from 2 and 4, and landscape 8 (loam; 299 µs cm^{-1}) could be separated from class 2. Landscape class 2 (warplands / fenlands) had the highest conductivity (769 µs cm^{-1}).

Comparisons for pH and conductivity of ditches were restricted to landscape classes 2, 4, 5 and 8. One-way ANOVA analysis showed that there were significant differences between landscape classes for pH ($F = 3.4$; $n = 47$; $p = 0.027$) and conductivity ($F = 27.6$; $n = 46$; $p < 0.001$). Tukey's HSD tests for pair-wise comparisons between landscape classes for pH showed that, although the ANOVA result for all landscape classes was significant, there were no individual landscape class pairs that were significantly different. It should be noted that Tukey's HSD is more conservative than a Student's t-test. The same pair-wise test applied to conductivity showed that landscape class 2 (warplands / fenlands) was significantly different ($p < 0.001$) from the other three landscape classes.

Although water pH varied significantly between landscape classes for all four water bodies, absolute differences in mean pH were relatively small. Mean pH varied by less than one unit across the different landscape for any single water body and by 1.5 pH units across landscapes and different water bodies. In general, the pH of ponds was lower than that of rivers, streams and ditches.

4. Discussion

For risk assessment to function correctly, it is essential that screening analyses are appropriately selected in order to distinguish low, intermediate and high risk situations and prioritise issues requiring more complex investigation. Preliminary assessment of exposure of aquatic ecosystems to pesticides in Europe relies on standard modelling scenarios (FOCUS, 2003). The data presented in this paper can be used to evaluate the scenarios for the areas considered. For example, British streams are generally wider than assumed in the standard scenarios (1 m) but are somewhat shallower than the assumption (0.3 m). There are two landscapes (clays and loams) where stream volume is on average smaller than the regulatory scenario, leading to a lesser potential for dilution of any pesticide loadings. The FOCUS scenarios assume that ponds exposed to pesticides have a total volume of 900 m^3. This is within the range of mean values for the twelve agricultural landscapes (461-1794 m3), but average volumes are smaller in four of the landscapes (fenlands, sandlands, eutrophic tills and loam landscapes).

Of the waterbodies examined, ditches were the most intimately related with agricultural land and ponds were the least. The regulatory scenarios assume that waterbodies are surrounded by agricultural land with only a 1- or 3-m margin to the nearest arable crop for ditches/streams and ponds, respectively. This assumption significantly overestimates the true proximity between water and crop. This is important because proximity influences both the level of contamination of water by pesticides and the potential for recovery of an impacted population through recolonisation from unaffected stretches of water. Impacts of pesticides on organisms in ponds may cause particular concern because of the relative isolation of these systems and the reduced potential for rapid recolonisation and recovery. However, data clearly show that most ponds are not directly proximate to arable land and that potential for direct impacts from pesticides in Britain is likely to be over-stated by current risk assessments.

Direct comparison of measured data with current assessment scenarios can help to place the assumptions into context. However, there is a need to compare exposure calculated using standard scenarios with the distribution of concentrations that results from considering the distribution of environmental conditions. Examples of such comparisons have been reported by Travis and Hendley (2001) and Brown et al. (2003) for the aquatic compartment and by Hart (2003) for birds. In each case, the screening-level estimate of pesticide exposure lay within the upper 5% of the distribution of exposure obtained using the range of measured data as input. Further work of this kind will help to quantify the level of protection afforded by screening assessments and ensure that modelling assumptions are appropriately selected.

The risk assessment carried out for non-target aquatic organisms is predominantly deterministic, taking single point estimates for both toxicity and exposure. The exposure value is often based on a point from a distribution so that, for example, deposition from spray drift is selected as the 90th percentile value from a database of measurements. The deterministic expression of risk is coupled with arbitrary safety factors and leads to a qualitative final output which tends to describe the risk in terms of 'margin of safety', 'adequate protection' or by reference to a higher tier study or studies. Such assessments do not provide an indication as to the magnitude or frequency of effects or to the level of certainty associated with the risk analysis. There is increasing interest in the use of probabilistic techniques within risk assessment for pesticides (Hart, 2001). These approaches explicitly quantify variability and uncertainty in the assessment and produce outputs with more ecological meaning, such as the probability and magnitude of effects. The analysis of information presented here is suitable for inclusion within probabilistic modelling of exposure with summary statistics to support definition of probability distribution functions or the potential to sample directly from the raw data. Inclusion of correlation between input parameters is an important consideration within probabilistic risk assessment (e.g. Cullen and Frey, 1998). To some extent the issue is reduced by the grouping of waterbodies into relatively homogeneous landscape classes. Relationships between parameters can be incorporated through detailed correlation analysis or by sampling individual water bodies into the analysis.

EU legislation on pesticides dictates that registration is only possible where "no unacceptable effects" on non-target aquatic organisms are expected to occur (EC, 1991). However, the Directive stops short of defining "acceptable" and "unacceptable" effects (Anon, 2002), even though a clear understanding of the protection target is a prerequisite for well-founded risk assessment (ref). One way to formalise this will be through the definition of reference images which describe the water bodies and their associated species assemblages which are to be considered in risk assessment (Giddings et al., 2002). Definition of reference images needs to be based on knowledge of regional variation in structure and function of aquatic ecosystems. Coupled with the analysis of the biota of British waterbodies described by Biggs et al. (200x), the data presented will help to inform the debate on what it is that we are trying to protect.

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Table 1. Definitions for the four waterbody types considered

Waterbody	*Definition*
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. Streams 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.

Table 2. Datasets used to describe the morphological and physico-chemical characteristics of agricultural aquatic habitats

Waterbody type	Surveys	No. of sites[1]	Data included in characterisation
Rivers and streams	Environment Agency River Habitat Survey (1994-96)	4500	20 channel and bank physical structure descriptors on 500 m survey lengths
	DEFRA Countryside Survey (2000)	420	20 channel and bank descriptors
	Environment Agency General Quality Assessments[2]		12 chemical parameters (pH, conductivity, BOD, COD, TON, NO_2, PO_4, Ca, Na, Mg, Cl, K)
Ponds	DETR Lowland Pond Survey (1996)	290	25 physical structure descriptors; 4 chemical parameters (pH, conductivity, alkalinity, Ca)
	Ponds Conservation Trust National Pond Survey (1989-98)	271	15 chemical parameters (pH, conductivity, Al, Zn, SS, Pb, Ni, Fe, Cu, TON, PO_4, Ca, Na, Mg, K)
Ditches	ADAS ditch surveys in Environmentally Sensitive Areas (ESAs) (1999)	1591	9 physical structure descriptors
	Ponds Conservation Trust targeted surveys (see Section 2.2.2)	48	3 physical structure descriptors; 7 chemical parameters (pH, conductivity, total P, NO_3, dissolved O, COD and BOD)

[1] Some parameters were not recorded from all sites, so site number varies for some parameters for certain surveys
[2] For chemical data, streams and rivers were not differentiated, so data jointly refer to both waterbody types

Table 3: Physical characterisation of British landscape classes

No.	Landscape	Description	Total area (km^2)	Dominant water bodies	Groundwater	Dominant water flow
1	River floodplains and low terraces	Level to very gently sloping river floodplains and low terraces	7,781	Rivers, streams, ponds & some ditches	Normally present at <2 m depth	Vertical
2	Warplands, fenlands and associated low terraces	Level, broad ‘flats’ with alluvial very fine sands, silts, clays and peat	9,017	Ditches and rivers	Normally present at <2 m depth	Vertical or saturated lateral
3	Sandlands	Level to moderately sloping, rolling hills & broad terraces. Sands and light loams	10,871	Rivers (and some ponds & streams) in low lying areas	Normally present at >2 m depth	Vertical
4	Till landscapes (eutrophic)	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	Ditches, streams, ponds & rivers	Generally none present	Predominantly saturated lateral
5	Till landscapes (oligotrophic)	Level to gently sloping glacial till plains. Medium loams and clays with low base status (oligotrophic). Some lighter textured soils on outwash	15,449	Ditches, streams, ponds & rivers	Generally none present	Predominantly saturated lateral
6	Pre-Quaternary clay landscapes	Level to gently sloping vales. Slowly permeable, clays (often calcareous) and heavy loams. High base status (eutrophic)	19,706	Ditches, streams, ponds & rivers	None present	Saturated lateral
7	Chalk and limestone plateaux and coombe valleys	Rolling ‘wolds’ & plateaux with ‘dry’ valleys. Shallow to moderately deep loams over chalk & limestone	14,197	Rivers, and possibly seasonal streams	Present at >2 m depth	Vertical

Table 3: continued…

No.	Landscape	Description	Total area (km^2)	Dominant water bodies	Groundwater	Dominant water flow
8	Pre-Quaternary loam landscapes	Gently to moderately sloping ridges & vales & plateaux. Deep, free-draining & moderately permeable silts & loams	10,072	Streams, ponds & rivers; possibly some ditches locally	None present	Saturated lateral
9	Mixed, hard, fissured rock and clay landscapes	Gently to moderately sloping hills, ridges and vales. Mod. deep free draining loams mixed with heavy loams and clays in vales	12,259	Streams and rivers with ponds in clay areas	Either none or present at >2 m	Saturated lateral; some vertical over groundwater
10	Hard rock landscapes	Gently to moderately sloping hills and valleys. Mod. deep free draining loams over hard rocks. Some slowly permeable heavy loams on lower slopes and valleys	23,342	Streams & rivers	None	Lateral along rock boundaries
11	SCOTLAND ONLY: Moundy morainic & fluvioglacial deposits	Gently & moderately sloping mounds, some terraces. Free draining morains, gravels & sands on mounds, poorly draining gleys in hollows	2,270	Streams & rivers	Variable	Vertical over groundwater; some saturated lateral
12	SCOTLAND ONLY: Footslopes with loamy drift	Concave slopes or depressional sites, often with springlines	1,081	Streams & rivers, occasional ditches		
13	Non-agricultural	All areas not cultivated with arable (including orchards, soft fruit and horticultural) or maintained grassland	79,690	Ditches, streams, ponds & rivers	Variable	Variable

Table 4. Number of sampling sites for different water bodies in the 13 landscapes (parentheses indicate that the water body is little found in that landscape)

No.	Landscape class	Number of sites [a]			
		Rivers	Streams	Ponds	Ditches
1	River floodplains and low terraces	2457 – 4926	153 – 306	28	259
2	Warplands, fenlands and associated low terraces	679 – 1360	98 – 196	11	1259
3	Sandlands	415 – 830	77 – 154	17	(5)
4	Till landscapes (eutrophic)	642 – 1284	241 – 482	32	30
5	Till landscapes (oligotrophic)	232 – 464	85 – 170	(6)	10
6	Pre-Quaternary clay landscapes	773 – 1548	348 – 696	55	11
7	Chalk and limestone plateaux and coombe valleys	307 – 614	96 – 192	20	0
8	Pre-Quaternary loam landscapes	421 – 842	319 – 638	22	9
9	Mixed, hard, fissured rock and clay landscapes	508 – 1016	109 – 218	16	(0)
10	Hard rock landscapes	818 – 1636	204 – 408	14	(0)
13	Non-agricultural	1123 – 2246	432 – 864	57	(8)

[a] The number of sites from which data was obtained varied according to the survey taken, and sometimes for specific parameters within each survey (i.e. certain parameters may have been included at some survey sites and not others); bankside properties were reported for each bank of rivers, streams and ditches.

Table 5: Agricultural land use across different landscape classes

No.	Landscape	Average field size (ha)	Average land use (% of total area)						
			Total agricultural	Cereals	Oilseed rape	Potatoes	Sugar beet	All fruit	Maintained grassland
1	River floodplains and low terraces	6.6	69.9	20.4	2.3	1.3	1.9	0.3	34.7
2	Warplands, fenlands and associated low terraces	23.9	80.8	33.5	2.6	3.7	6.1	0.4	18.4
3	Sandlands	5.5	65.7	20.6	1.2	2.3	4.4	0.3	27.8
4	Till landscapes (eutrophic)	7.2	76.1	31.2	3.5	1.1	2.1	0.2	27.4
5	Till landscapes (oligotrophic)	3.9	67.0	11.3	1.4	0.3	0.1	<0.1	50.5
6	Pre-Quaternary clay landscapes	5.1	70.8	22.4	3.2	0.6	0.7	0.5	32.7
7	Chalk and limestone plateaux and coombe valleys	8.7	73.7	33.9	4.1	0.7	1.4	0.3	19.6
8	Pre-Quaternary loam landscapes	4.1	66.3	20.9	1.9	1.0	0.6	1.6	30.4
9	Mixed, hard, fissured rock and clay landscapes	2.8	67.1	7.5	0.5	0.3	0.1	<0.1	56.0
10	Hard rock landscapes	2.7	62.4	4.8	0.2	0.4	<0.1	<0.1	54.4

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

Figure 1. Distribution of British landscape classes

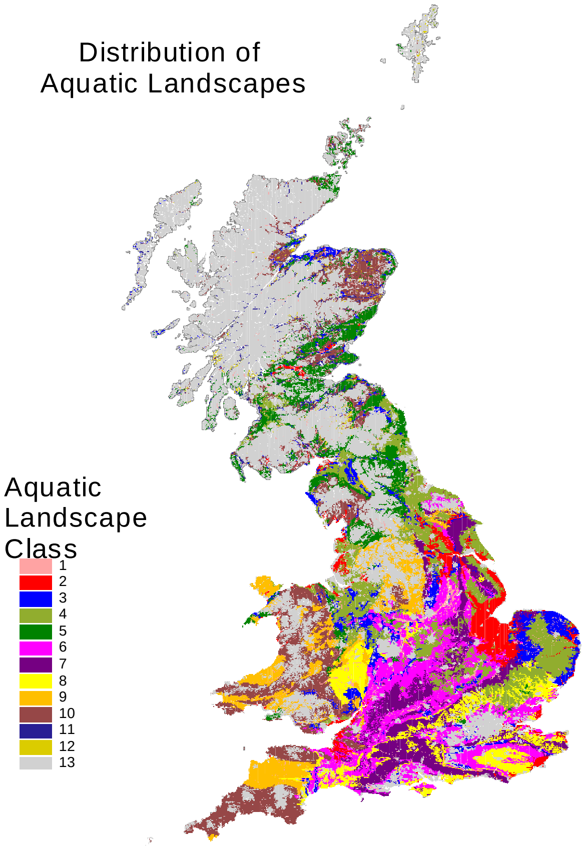


Figure 2. Length or number of water bodies per unit area of the different landscapes

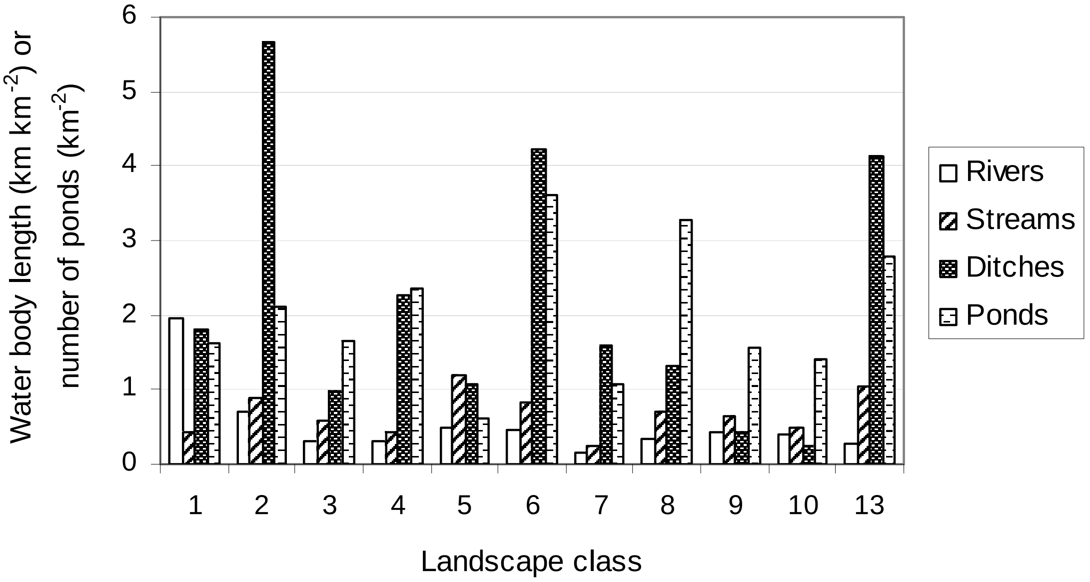


Figure 3. Major divisions of land use for the different landscapes in England and Wales

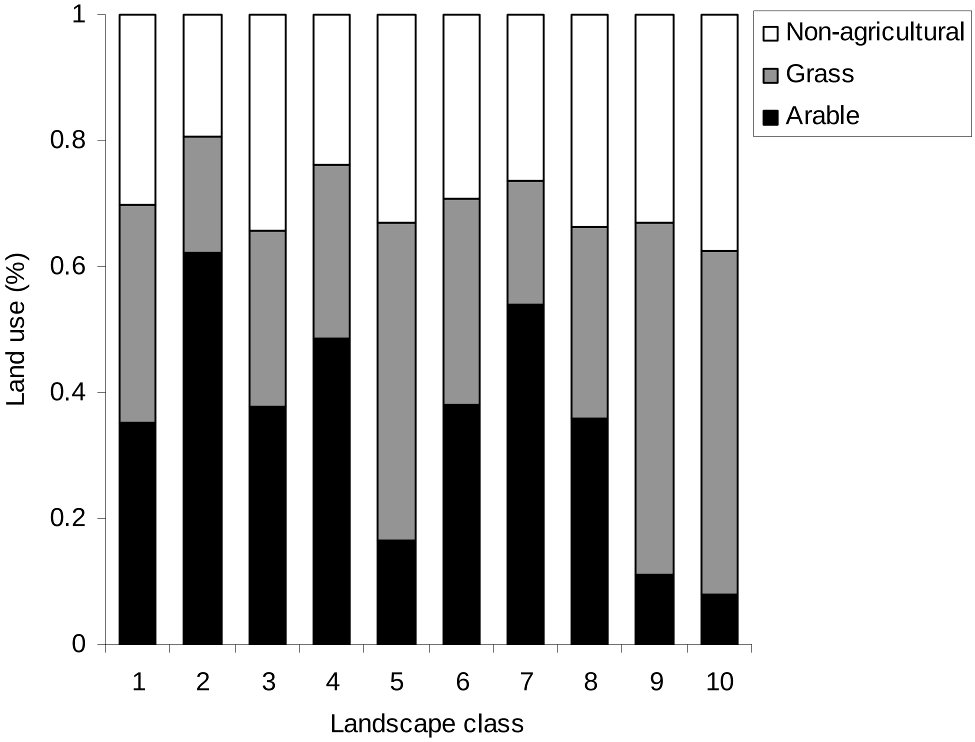
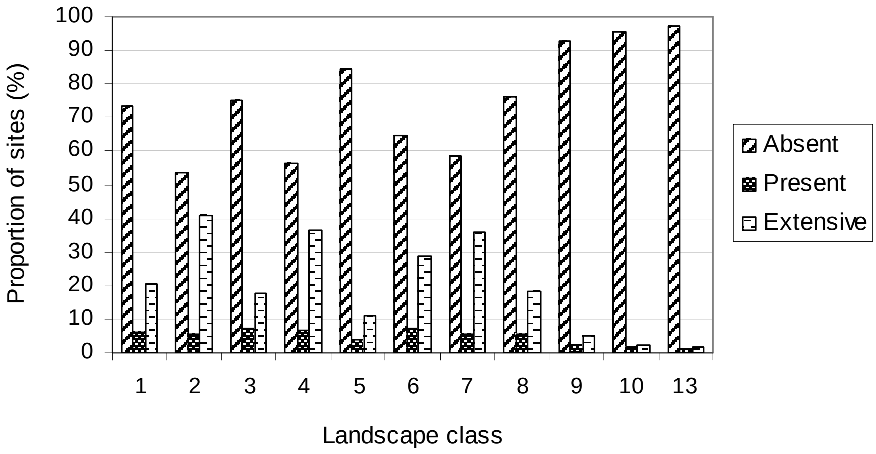
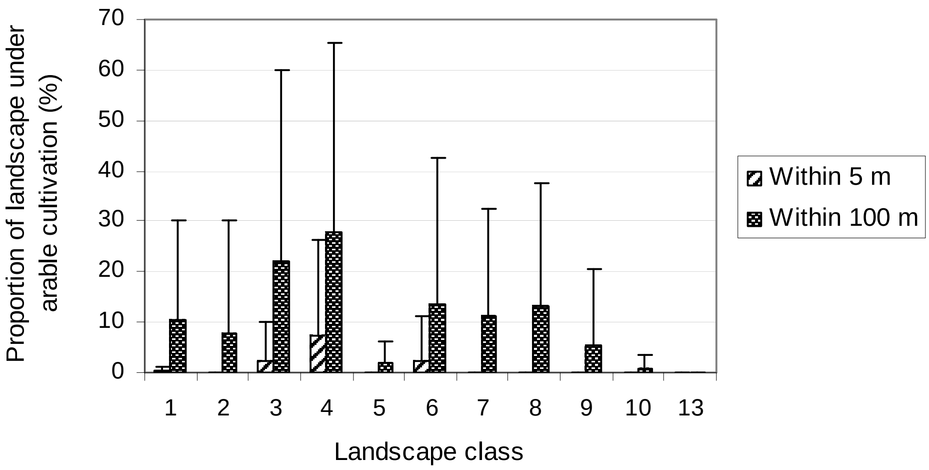


Figure 4. Measures of the density of arable cultivation around a) rivers and streams and b) ponds, or c) of the distance between ditches and the nearest arable field

a)



b)



c)

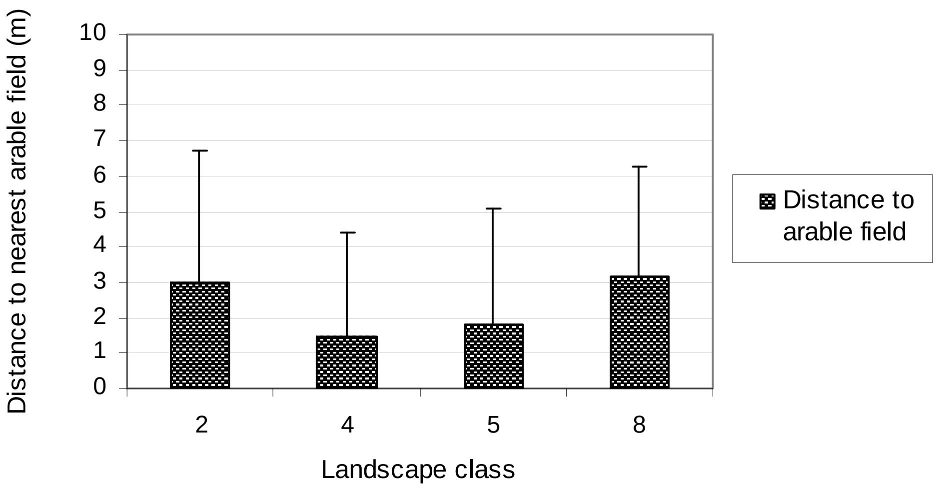


Figure 5. Ordering of streams and rivers in different landscape classes according to a) width, b) depth and c) shape parameter.

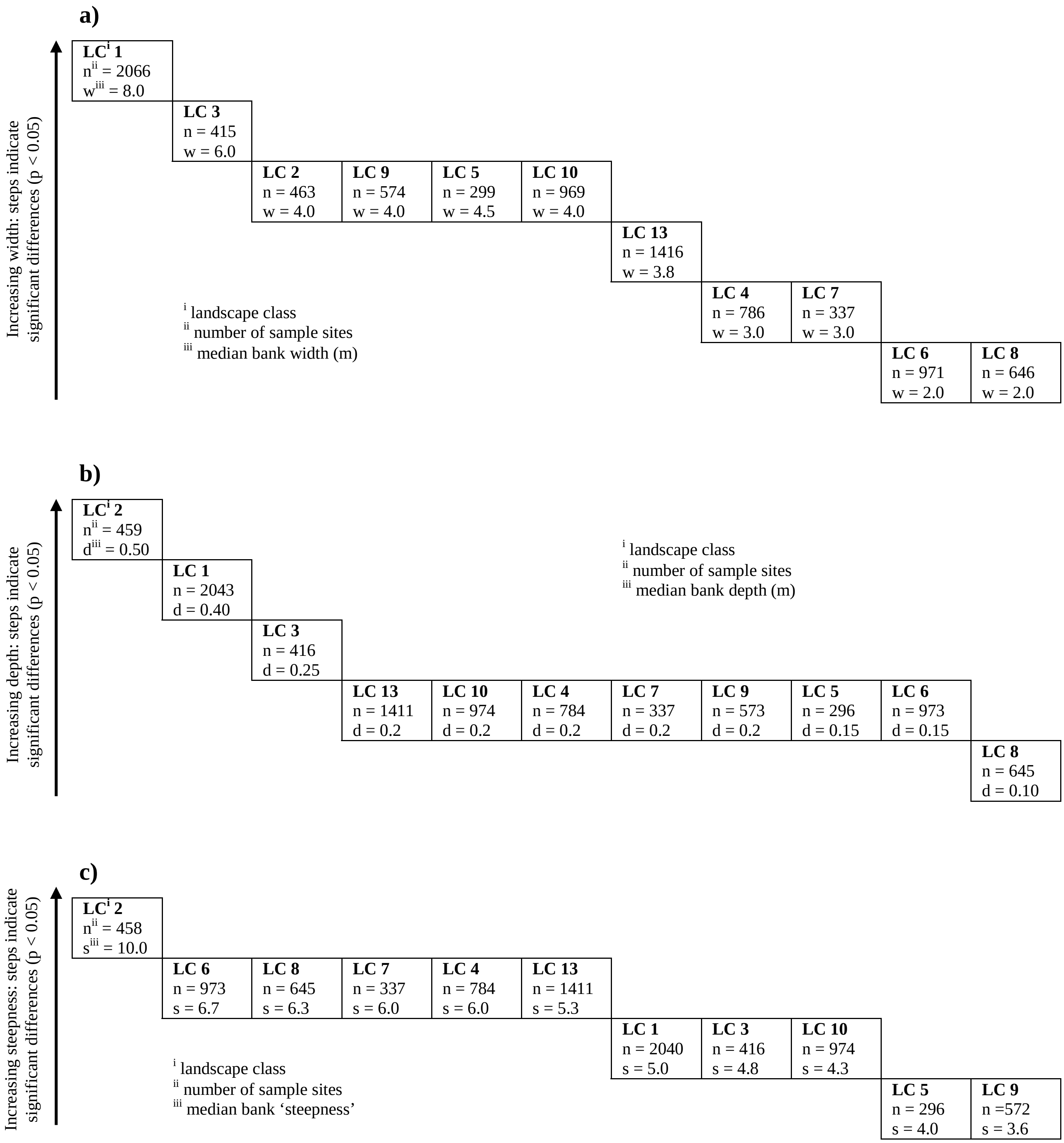


Figure 6. The categorical distribution of ditch widths between landscape classes 1, 2, 4 and 6

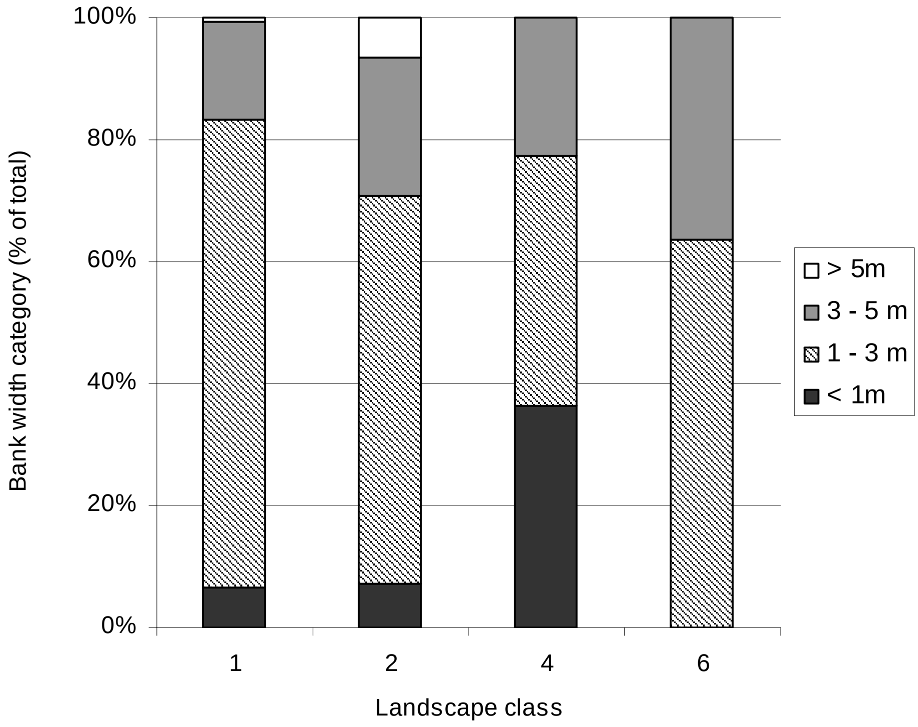


Figure 7. Bed material in the streams and rivers of different landscape classes

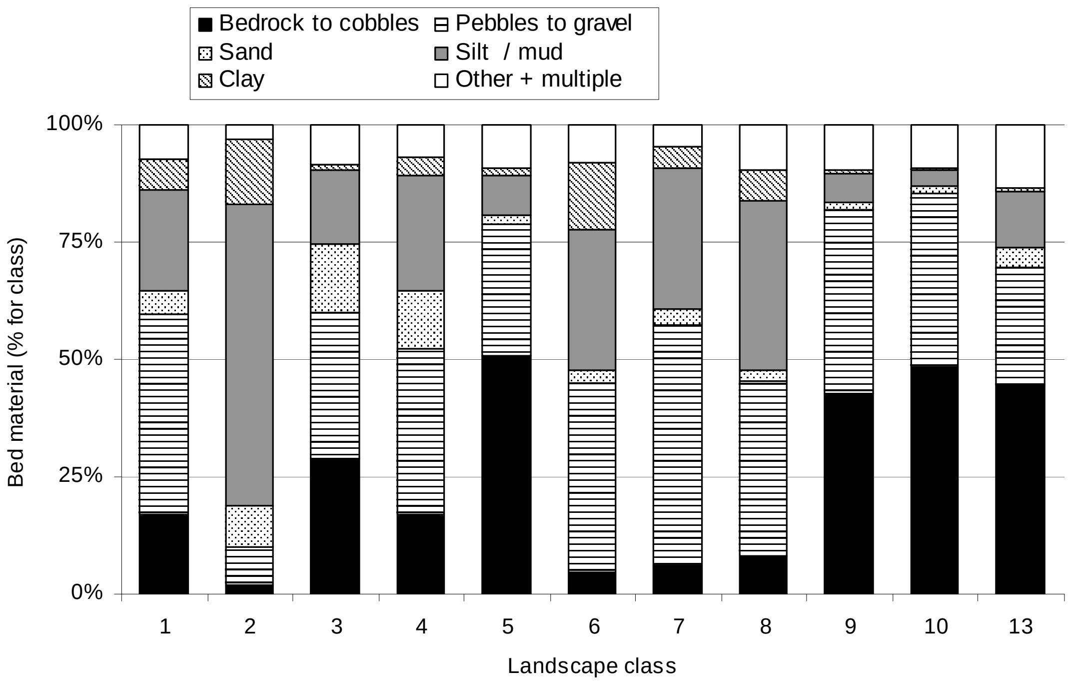


Figure 8. Mean pH values for different waterbodies in the different landscapes (error bars are sample standard deviations)

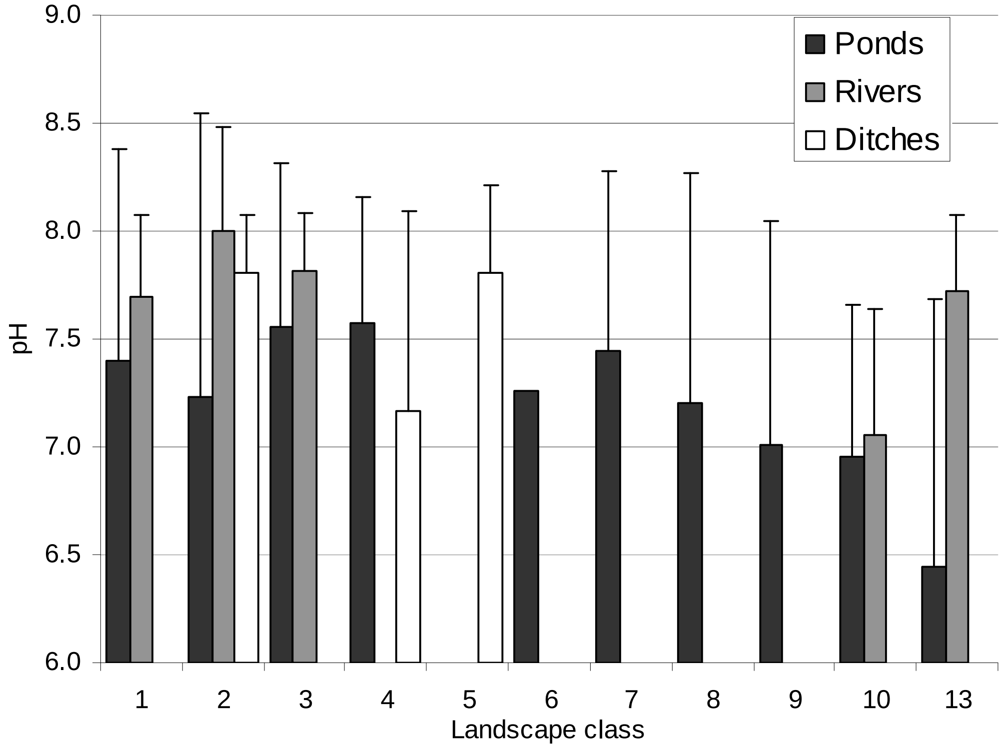
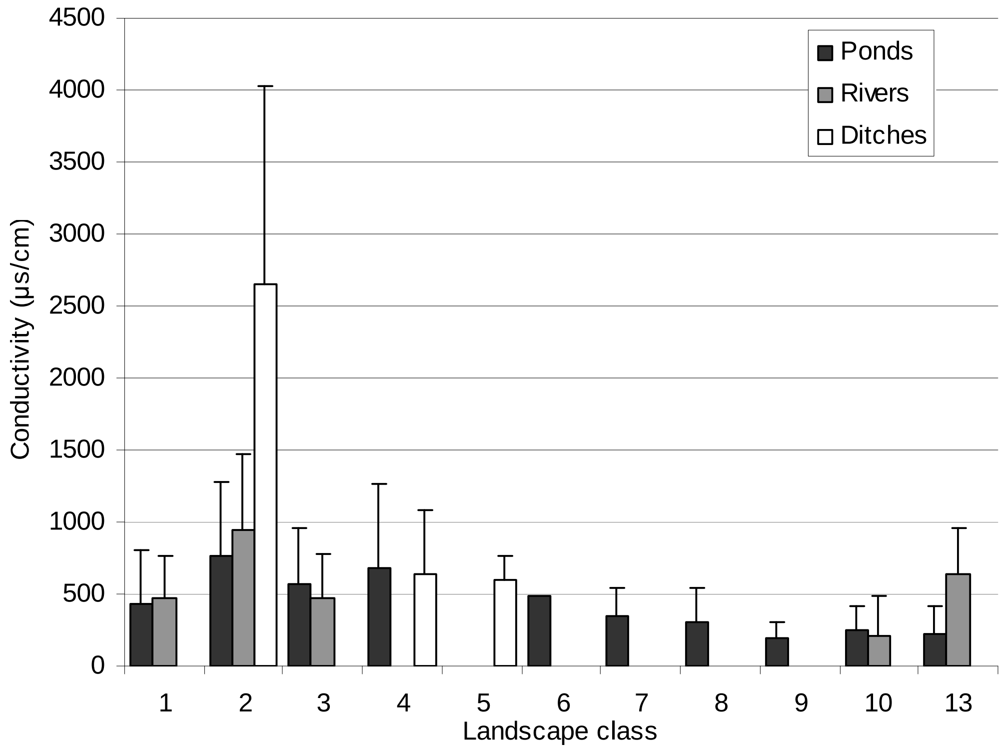


Figure 9. Mean conductivity values for different waterbodies in the different landscapes (error bars are sample standard deviations)



Morphological and physico-chemical properties of British aquatic habitats potentially exposed to pesticides.

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