Hydrobiologia (2008) 597:7–17
DOI 10.1007/s10750-007-9227-6

A comparison of the catchment sizes of rivers, streams, ponds, ditches and lakes: implications for protecting aquatic biodiversity in an agricultural landscape

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Abstract In this study we compared the biodiversity of five waterbody types (ditches, lakes, ponds, rivers and streams) within an agricultural study area in lowland England to assess their relative contribution to the plant and macroinvertebrate species richness and rarity of the region. We used a Geographical Information System (GIS) to compare the catchment areas and landuse composition for each of these waterbody types to assess the feasibility of deintensifying land to levels identified in the literature as acceptable for aquatic biota. Ponds supported the highest number of species and had the highest index of species rarity across the study area. Catchment areas associated with the different waterbody types differed significantly, with rivers having the largest average catchment sizes and ponds the smallest. The important contribution made to regional aquatic biodiversity by small waterbodies and in particular ponds, combined with their characteristically small catchment areas, means that they are amongst the most valuable, and potentially amongst the easiest, of waterbody types to protect. Given the limited area of land that may be available for the protection of aquatic biodiversity in agricultural landscapes, the deintensification of such small catchments (which can be termed microcatchments) could be an important addition to the measures used to protect aquatic biodiversity, enabling 'pockets' of high aquatic biodiversity to occur within working agricultural landscapes.

Keywords Watershed · Microcatchment · Aquatic biodiversity · Agri-environment schemes · Diffuse pollution

Guest editors: R. Céréghino, J. Biggs, B. Oertli & S. Declerck
The ecology of European ponds: defining the characteristics of a neglected freshwater habitat

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Introduction

Pollution from agriculture is recognised as having a significant negative impact on water quality and aquatic biota (Allan, 2004; Foley et al., 2005; Declerck et al., 2006; Donald & Evans, 2006). These pollutants include nutrients and other chemicals used to maximise production on arable land; animal waste and animal health biproducts, e.g. antibiotics and sheep dip, from pastoral land; as well as sediment resulting from eroded soils. They affect aquatic ecosystems both by altering the physicochemical characteristics and quality of a waterbody (e.g. eutrophication, changes to sediment composition) and by direct toxicity impacts on the organisms within it. Many of these pollutants are diffuse in nature, and the broad areas from which they emanate and multiplicity of pathways by which they reach waterbodies, make such pollutants difficult to control and mitigate.

The quantity and concentration of diffuse pollutants reaching waterbodies can potentially be reduced by two mechanisms: (i) source control through reduction in chemical loads, better targeting in the timing of chemical applications and the use of appropriate farming techniques to reduce runoff, e.g. minimum tillage; and (ii) measures to prevent pollutants from reaching waterbodies through deintensifying areas of land, e.g. buffer zones. Although the source control mechanisms go someway towards reducing pollution (e.g. Yates et al., 2006) diffuse pollutants will not be completely eliminated by such means. Thus, methods of land management and pollutant interception are likely to remain important in the effective long-term protection of aquatic biota under current agricultural systems. Typically such methods involve leaving areas adjacent to waterbodies out of agricultural production, through whole field conversion or more commonly, the creation of buffer strips. Although widely used and much tested, this approach has shown mixed results in terms of pollution reduction, e.g. Schmitt et al. (1999), Dosskey (2002), Borin et al. (2004, 2005), and recent evidence (e.g. Wang et al., 1997; Quinn, 2000; Fitzpatrick et al., 2001; Donohue et al., 2006) implies that where such methods have proved relatively ineffective, this has often been because the deintensified area has not included a sufficient proportion of the catchment area of a waterbody.

The catchment of a waterbody is the area over which water, and hence diffuse pollutants, will travel (both by overland and subsurface movement) to enter the waterbody. Catchments have long been recognised as the key to understanding the ecology of freshwaters (Hynes, 1975; Allan et al., 1997; Allan, 2004) and although underlying geology and morphology are the fundamental determinants of water characteristics (Host et al., 1997; Johnson et al., 2004; Wiley et al., 1997; McRae et al., 2004), in areas where landcover is heavily modified, the landuse composition of the catchment will dominate (Hynes, 1975; Lund & Reynolds, 1982; Moss et al., 1996; Allan et al., 1997; Muir, 1999; Cresser et al., 2000; Johnson & Goedkoop, 2002; Tong & Chen, 2002). Thus, the proportion of intensive landuse in a catchment as well as the catchment's size will influence the cost and potential success of a waterbody's protection from diffuse pollution.

Catchment areas are generally perceived as large and are usually described only in the context of rivers or large lakes. For example, in the UK, small river and stream catchments are generally termed 'sub-catchments'. However, in reality all waterbodies, large or small, have a catchment area. The association of catchment areas with larger rivers and lakes is likely to have resulted from the historic use of these waterbodies for navigation, drainage, food supply, water supply, recreation and removal of wastes. This considerable socio-economic value has resulted in a vested interest in the protection of these waterbodies and consequently, both scientific research and environmental protection has tended to be focused at larger waterbodies. The more limited economic value of small waterbodies has meant that until recently, their biodiversity potential has tended to be overlooked with a general presumption that they are inferior versions of their larger equivalents. However, recent evidence has shown that small waterbodies may in fact make a disproportionately large contribution to aquatic biodiversity across landscapes in terms of both their species richness and their species rarity (Biggs et al., 2003, 2007; Williams et al., 2004; De Bie et al., 2008; Davies et al., in press), implying that they are likely to warrant a higher priority in terms of conservation concern. The ease and success of their protection from diffuse agricultural pollution will depend to some extent, as for larger waterbodies, on the proportion of their catchment areas that can be incorporated into protection strategies.

This study investigates the aquatic biodiversity (species richness and rarity) of a suite of waterbody types across an area of UK lowland agricultural landscape in the context of their catchment sizes. The results are used to explore the potential ease and success of the protection of different waterbody types from diffuse agricultural pollution.

Material and methods

The study area and its aquatic biodiversity

A 13 × 11 km study area of lowland agricultural landscape in Britain on the borders of Oxfordshire, Wiltshire and Gloucestershire was selected. The study area contained three Department for Environment, Food and Rural Affairs (Defra) agricultural landscape classes (Table 1) (Biggs et al., 2003) and was considered typical of lowland agricultural landscapes (Brown et al., 2006). Arable cultivation dominated the landcover (75%), with 9% under woodland, 7% improved grassland, 2% urban and the remaining 7% made up of water, semi-natural grassland and bare rock (Fig. 1). Agricultural land was predominantly arable, comprised mainly of cereals, permanent grass, oil-seed rape, potatoes peas and sugarbeet. There were 205 ha of surface water in the study area comprising 3 rivers, 97 streams, 236 ponds, 8 lakes and 340 ditches. The rivers included lengths of the Thames (c. 16.7 km), Cole (c. 16.8 km) and Coln (c. 4.3 km).

Data on the macrophyte and macroinvertebrate species present in the five dominant waterbody types in the study area (ditches, lakes, ponds, rivers and streams) were collected in two phases. In 2000, a stratified random sample of 20 sites was surveyed for each of four waterbody types (ditches, ponds, rivers and streams), i.e. 80 sites in total (reported in Williams et al., 2004). During 2002–2003, comparable data were collected from a further 20 sites in a fifth waterbody type, lakes. Due to the size division between a lake and a pond (Table 2; Biggs et al., 2003; Williams et al., 2004), data were also collected for a pond to replace one from the existing dataset which would have been categorised as a small lake under Table 2.

Within each waterbody, the sample area and survey methods followed those used by Williams et al. (2004). Each sample area was 75 m^2 to enable direct comparison of data from different waterbody types with inherently different sizes. For linear waterbodies, this area comprised a rectangular section of the waterbody, while for circular waterbodies, the area comprised a triangular wedge with the base following the margin and the apex at the centre of the waterbody. At each site, all wetland macrophytes (marginal, emergent, submerged and floating-leaved plants) were recorded by walking and wading the margin, using a grapnel thrown from the bank and sampling from a boat in the deeper lake sites. A three-minute hand net sample was taken for macroinvertebrates using a standard 1 mm mesh net, with the three minute sample time being divided equally between the mesohabitats in the 75 m^2 area. Macroinvertebrate samples were exhaustively live-sorted in the lab and all individuals (except Diptera larvae and Oligochaeta which were omitted from the analysis) were identified to species level, except very abundant taxa (>100 individuals) which were sub-sampled.

The macrophyte and macroinvertebrate data provided information on aquatic species richness and rarity for each survey site in the ditches, lakes, ponds, rivers and streams (100 sites in total). The species richness of a site was the total number of species found at that site, whilst species rarity was calculated using the Species Rarity Index (SRI). This rarity index follows a process developed by Foster et al. (1990) whereby each species is given a numerical value according to its rarity or threat within Britain, the total for each site is then summed and finally divided by the number of species found at the site, resulting in an index which is not biased towards

Table 1 Defra agricultural landscape classes occurring within the study area

Landscape	Area-km^2 (% of area)	Description	Associated agriculture
LC1—River floodplains and low terraces	16.27 (11.5)	Level to very gently sloping river floodplains and low terraces	Permanent grass, some cereals and oil-seed rape, probably more intensive on terraces
LC6—Pre-quaternary clay landscapes	95.46 (67.1)	Level to gently sloping vales. Slowly permeable, clays (often calcareous) and heavy loams. High base status (Eutrophic)	Permanent grass, cereals (>10–15%), leys, oil-seed rape maize and beans
LC7—Chalk and limestone plateaux and coombe valleys	30.44 (21.4)	Rolling 'Wolds' and plateaux with 'dry' valleys; shallow to moderately deep loams over chalk and limestone	Cereals (and oil-seed rape, beans), sugar beet, potatoes, peas

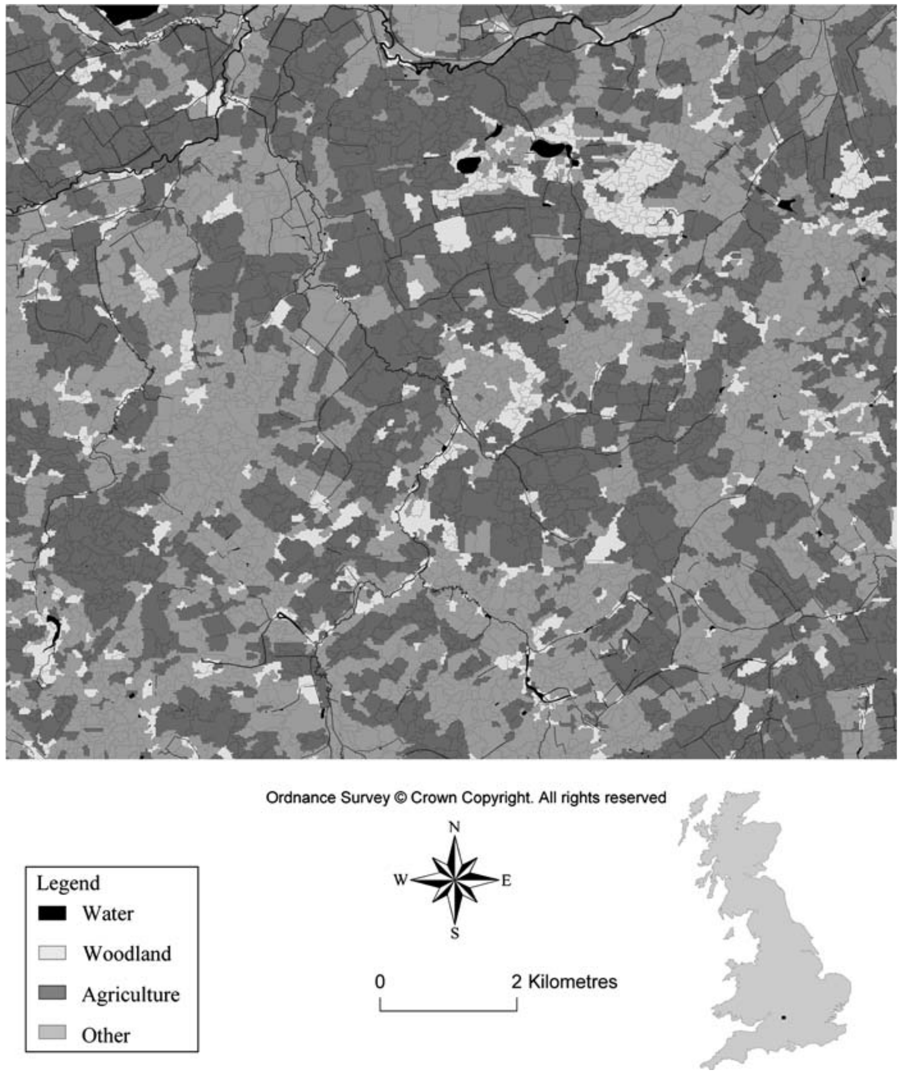

Fig. 1 The study area and its landuse composition

Table 2 Definitions of waterbodies used in this study

Waterbody type	Definition
Lakes	Bodies of water, both natural and man-made, greater than 2 ha in area (Johnes et al., 1994). Includes reservoirs, gravel pits, meres and broads.
Ponds	A body of water, both natural and man-made, between 25 m^2 and 2 ha in area, which may be permanent or seasonal (Collinson et al., 1995).
Rivers	Relatively large lotic waterbodies, created 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.
Streams	Relatively small lotic waterbodies, created by natural processes. Marked as a single blue line on 1:25,000 OS maps and defined by the OS as being less than 8.25 m in width. Streams differ from ditches by usually: (i) having a sinuous planform; (ii) not following field boundaries; and (iii) showing a relationship with natural landscape contours, usually by running down valleys.
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.

species-rich sites (see Williams et al., 2004 for further details).

Catchment delineation and landcover

Ordnance Survey (OS) Landform Profile and MasterMap data were used to create the underlying Digital Elevation Model (DEM) upon which catchment delineation was based, for an area extending 8 km outside the study area, using the Geographical Information System (GIS) software, ArcGIS 8.2. MasterMap data include topographic information on every landscape feature, each with its own unique identifying code. Landform Profile data comprise height data at 10 m intervals in x and y and recorded to the nearest 0.1 m in z, with a planimetric accuracy of ± 1 m and a vertical accuracy of ± 1.8 m.

All waterbody polygons were extracted from the MasterMap data. Misclassified features were removed and polygons split by overlying features, such as bridges, were joined. Separate data layers were created for ditches, lakes, ponds, rivers and streams according to the definitions in Table 2. A new river or stream was defined at confluence sites and a new ditch was defined at both confluence sites and where it turned by approximately 90°. Results were visually compared with 1:25,000 OS maps, aerial photographs and site visits to ensure that the network of catchments and waterbodies generated from digital data were consistent with the real landscape.

Each waterbody polygon was assigned a constant minimum height value from the OS Landform Profile data and the waterbodies layer converted to a 5 m grid. The 10 m Landform Profile raster data were also converted to a 5 m grid using bilinear resampling, so that it could be combined with the waterbodies whilst retaining their continuity. The waterbodies were 'burnt' into the DEM at their height value minus 10 m to ensure that modelled runoff would flow into the waterbodies and that they would be retained during the catchment delineation process.

The catchments of each waterbody type were delineated separately using the DEM with the depressed waterbodies and the ArcGIS extension ArcHydro Tools (Version 1.1 Beta 2; ESRI, 2001). ArcHydro Tools only modelled surface water flow. Although the study area was predominantly underlain by impermeable clay and so overland flow would have been a dominant process, some throughflow and transport via field drains would have occurred but this was not modelled and is a limitation of the method. ArcHydro Tools also had the underlying assumption that all water will flow to the edge of the DEM. This ignores standing waterbodies which provide natural sinks that retain water within a landscape and so all ponds and lakes were 'seeded' with 'no-data' points at their deepest locations or centre point, to ensure that water flowed towards these points.

River catchments that extended beyond the limit of the data held were estimated using published statistics on catchment size from gauging stations present in the study area (Environment Agency, 2006). Additional manipulation of one river catchment boundary was required by hand due to the very flat nature of the northwest corner of the study area, which meant that ArcHydro Tools was not able to accurately define the catchment boundary in this instance. The landuse composition of the catchments was ascertained by intersecting each catchment with Land Cover Map 2000 data (remotely sensed landuse data in 25 m^2 cells; copyright NERC). For the river catchments that extended beyond the limit of data available, the proportions of different landuse types were taken as those modelled in the GIS for the area over which data was held.

Results

Of the total area of water (205 ha) within the study area (13 × 11 km), lakes comprised the greatest proportion of the water area (38%), followed by rivers (24%), ditches (20%), streams (14%) and ponds (4%) (Fig. 2). There was a broadly inverse relationship between surface water area and number of waterbodies, with rivers and lakes being fewest in number but covering the largest surface area and ponds being one of the most numerous waterbody types but having the smallest total surface area.

Across the study area, the pond sites supported the greatest number of both macrophyte and macroinvertebrate species, followed by rivers, lakes, streams and lastly ditches, which contained no aquatic (submerged or floating) plants (Fig. 3a). Overall, the pond sites supported 238 species, river sites

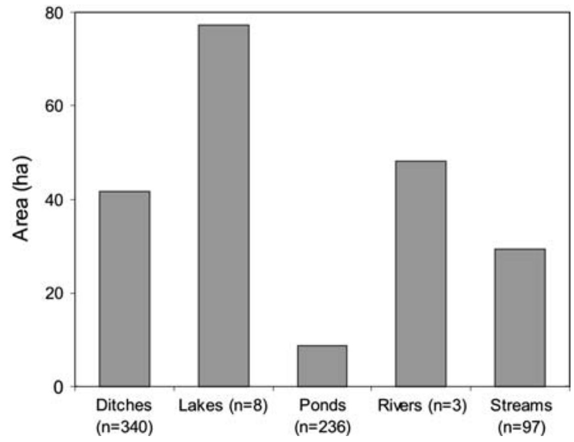

Fig. 2 Surface water area of the different waterbody types within the study area

supported 201 species, lakes 186 species, streams 163 species and finally, ditch sites supported 120 species.

The pond sites had the highest SRI for macrophytes, followed by lakes, rivers, streams and lastly ditches which supported no rare plant species (Fig. 3 b). The ponds also had the highest macroinvertebrate SRI, followed by ditches, lakes, streams and lastly rivers (Fig. 3b).

Catchment sizes were significantly different between waterbody types (Kruskall–Wallis, $P < 0.001$). Rivers had the greatest average catchment areas (43,850 ha), followed by lakes (141 ha), streams (86 ha), ditches (29 ha) and lastly, ponds (18 ha) (Fig. 4). The total catchment areas followed a different pattern: overall, rivers had the greatest total catchment area (131,550 ha), followed by ditches (9,904 ha), streams (8,354 ha), ponds (4,237 ha) and lastly, lakes (1,124 ha) (Fig. 4). This difference arose because ditches had catchments that were small in size but numerous giving a large total catchment area, whilst lakes had large catchments but were few in number. Although river catchments were clearly the largest, there were too few in the sample for this to be confirmed with post hoc Mann–Whitney *U* tests. However, significant differences ($P < 0.001$) in catchment size were seen between ponds and streams, ponds and lakes, ditches and streams and ditches and lakes, whilst no significant difference was observed between the size of pond and ditch catchments and between stream and lake catchments. This showed that ponds and ditches had similarly small catchments, whilst streams and lakes had similarly larger catchments and rivers had the greatest catchment sizes.

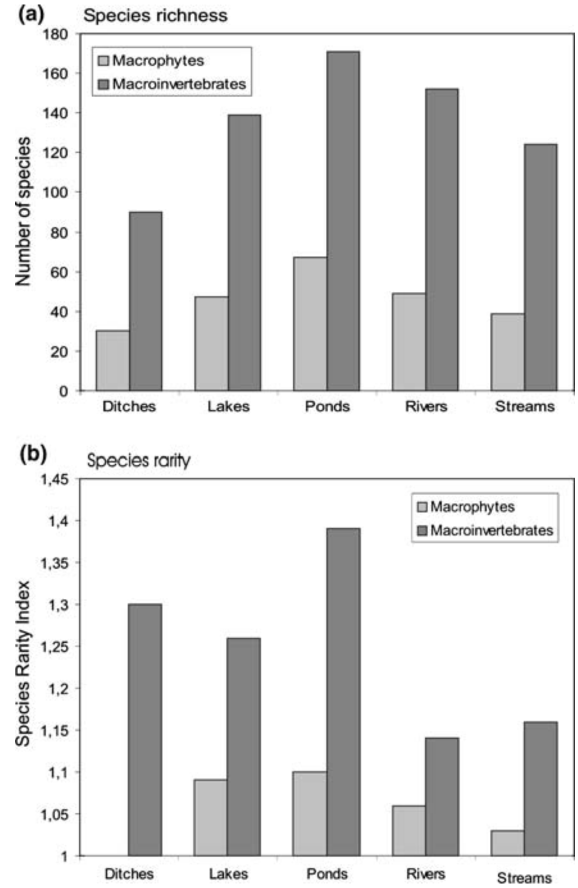

Fig. 3 Macrophyte and macroinvertebrate (**a**) species richness and (**b**) species rarity across the study area

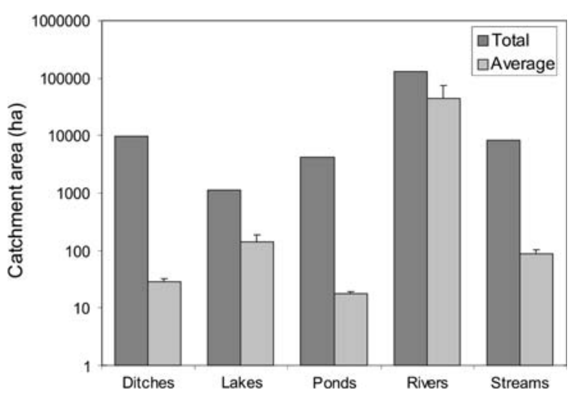

Fig. 4 Average and total catchment areas of the different waterbody types within the study area

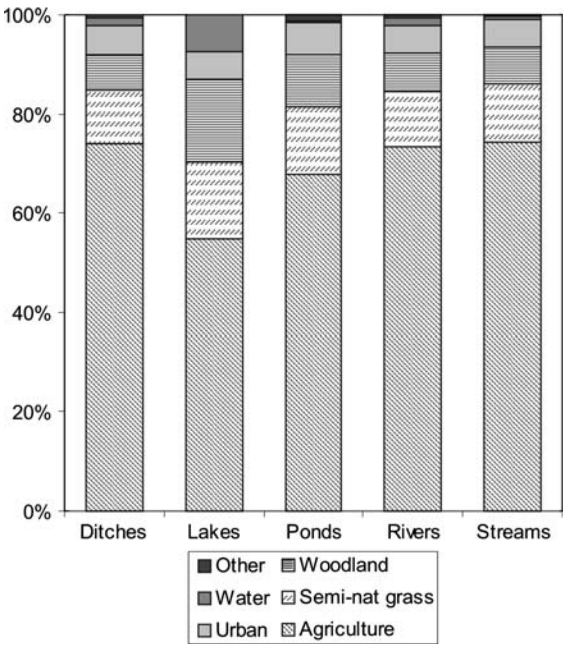

Fig. 5 Proportion of landuses within the total catchment area of each waterbody type

The catchments of all waterbody types were dominated by agricultural landuse, with river, stream, pond and ditch catchments all containing similar proportions of agricultural land (69–74%) together with similar proportions of woodland (7–10%), semi-natural grassland (10–13%) and water (1–2%) (Fig. 5). Lake catchments differed, typically containing less agricultural land (55%) and larger proportions of semi-natural grassland (c. 4% more), woodland (c. 9% more) and water (c. 6% more) than those of the other waterbody types. The proportion of urban land within catchments was similar for all waterbody types (c. 5%).

Discussion

Catchment sizes and the area needed for protection of aquatic biota

Hynes (1975), in his classic work on the ecology of running waters, described rivers as, "*a manifestation of the landscapes that they drain*". Since this pioneering work, many studies have sought to characterise river, stream and sometimes lake catchments, analysing relationships between factors such as catchment area, landuse composition and waterbody ecology. Practically all of these studies have reinforced Hynes' original comments. The extent of agriculture in the developed world is such that it is a dominant component of many waterbodies' catchment areas, with its associated diffuse pollutants having well accepted and largely detrimental impacts on aquatic biodiversity.

Recent research has investigated the importance of such widespread agriculture within catchments, typically finding streams to remain in good condition until agriculture exceeds 30–50% of the catchment (Allan, 2004). For example, working in the US, Wang et al. (1997) found that habitat quality and scores of biotic integrity declined when agricultural landuse exceeded 50% of the catchment. Fitzpatrick et al. (2001) working in the same region, found declines in fish biotic indices above 30% agricultural land, whilst Quinn (2000), working in New Zealand, found 30% agricultural land to be a critical value for macroinvertebrates. More stringent levels were identified by Donohue et al. (2006) who investigated the relationship between the ecological quality of aquatic networks and landcover, identifying thresholds at which 'good' ecological status could be attained in Ireland. They predicted that with more than 1.3% arable land in a catchment or 37.7% pastoral land, ecological status would fall below 'good' levels. Within the current study area, the average landuse composition of the catchments of all the waterbody types exceeded these thresholds. The waterbody type with the least intensive catchment landuse composition was lakes, which were, on average, associated with 55% agricultural land. This implied that within the study area, lakes were already fairly well buffered, largely due to their frequent location on large private estates. This may not, of course, be the situation in other areas.

Agriculture currently covers approximately half of the earth's habitable surface (Clay, 2004) and in many countries covers more than 70% of the land surface. This implies that vast areas would need to be deintensified to reach the maximum thresholds of 30–50% identified as important in the literature. Given the anticipated doubling of food demand forecast for the next 50 years (Donald & Evans, 2006), this may be very difficult to achieve and impractical in landscapes where agricultural production is needed. However, not all waterbodies have large catchment areas and it may be possible to deintensify those with 'microcatchments' to reach the critical thresholds of 30–50%, or even to completely deintensify them. The present study found that, as might be expected, larger waterbodies, (i.e. rivers and lakes), had larger catchments than smaller ponds, streams and ditches. Using the study area as an example, rivers would require a comparatively large amount of land to be deintensified to reach the thresholds of 30 and 50%, compared with the smallest waterbody type, ponds (Table 3). To attain levels of no more than 50% agricultural land in an average catchment, rivers would require 10,086 ha to be deintensified, compared with just 4 ha for ponds. To reach 30% agricultural land, rivers would require 18,856 ha to be deintensified, compared with just 7 ha for ponds.

Therefore, if ponds and other waterbodies with small catchments can be demonstrated to support high levels of biodiversity they may play an important part in a strategy for the protection of aquatic biodiversity because they could be afforded very high levels of protection, e.g. complete deintensification, for a comparatively low land area.

Macrophyte and macroinvertebrate species richness and rarity across the study area

Ponds made a high contribution to both the aquatic macrophyte and macroinvertebrate biodiversity of the study area in terms of both species richness and species rarity. Results for the other waterbody types were more mixed, with rivers supporting a relatively large number of species but low species rarity, whilst ditches supported few species but had a high macroinvertebrate SRI. Studies comparing the aquatic biodiversity of different types of waterbody are few and, as far as the authors are aware, none have compared the range of waterbody types undertaken for this study. However, those studies that have compared aquatic biodiversity for a more limited range of habitats have often found smaller waterbodies, particularly ponds and ditches, to make an important contribution (e.g. Painter, 1999; Armitage et al., 2003; Biggs et al., 2007; Davies et al., in press). These small waterbody types have often been overlooked in biodiversity protection and rarely enjoy the statutory protection afforded to larger waterbodies. The results of this study, supported by other work on comparative biodiversity including smaller waterbody types (Davies, unpublished data; Davies et al., in press), suggest that this may be both a considerable oversight and a missed opportunity. In particular, the valuable contribution of smaller waterbodies to regional aquatic biodiversity means that they could have an important role in the strategic protection of aquatic biota.

Table 3 Areas requiring deintensifying within the study area to reach identified thresholds of 50% and 30% agricultural land

Waterbody type	Average area (ha) requiring deintensification to reach:	
	50% Agricultural land	30% Agricultural land
Rivers	10085.5	18855.5
Streams	20.6	37.8
Lakes	7.1	35.3
Ditches	6.7	12.8
Ponds	3.6	7.2

The potential role for small waterbodies in protection strategies

As identified above, waterbodies with larger catchments require a much greater area to be deintensified to reach the levels that are suggested as needed for the sufficient protection of aquatic biota. In the study area, the largest catchments (associated with rivers) were on average 300 times larger than those of lakes (which had the second largest catchments) and almost 2,500 times larger than those of ponds, which had the smallest mean catchment sizes. Equally, in the area required to deintensify a single average-sized river catchment to 50% agricultural landuse, it would be possible to fully deintensify more than 560 average-sized pond catchments. Thus, the relatively small catchment sizes of smaller waterbodies, and in particular ponds, combined with their important contribution to aquatic biodiversity, means that their inclusion amongst the measures used for biodiversity protection from diffuse agricultural pollution is likely to enhance the effectiveness and economic efficiency of protection across whole landscapes.

The common perception, mentioned above, that catchment areas are associated with larger waterbodies and in particular rivers (and hence cover large areas), may be one of the reasons that whole catchment or large-scale deintensification are not generally proposed as protection measures. Instead, buffer strips (involving much smaller areas) are often employed. The disadvantage of such methods for larger waterbodies is that, depending on the proportion of the catchment that they occupy, they are unlikely to provide sufficient protection for the aquatic biota of the waterbody. For example, under the English Agri-Environment Scheme, Environmental Stewardship, the maximum buffer width offered at the edge of a field to protect a river is six metres. Published data on the effectiveness of buffer zones at varying widths suggest that a six metre buffer is highly unlikely to result in reduced nutrient pollution loads to rivers. Due to the large edge length of a river, the agricultural land-take that would be involved in even a limited buffer area would be considerable, implying that large areas of agricultural land are likely to be taken out of production to protect a river, with very little chemical or ecological gain.

The mechanisms most likely to be used to protect aquatic biodiversity in agricultural landscapes in the UK, are agri-environment schemes (AESs) whose structure would facilitate small catchment deintensification. These schemes remunerate farmers for environmentally sensitive farming, including reducing chemical and nutrient inputs as well as land management to reduce the impacts of diffuse pollution. The scheme is voluntary and is applied to by individual farmers and consequently, the measures offered are at a farm-scale. Such a scale and the potentially ad hoc uptake would favour microcatchment deintensification over that of larger catchments which would require the efforts of many farmers to be coordinated, increasing administrative costs and the complexity of operation, as well as increasing the chances of failure because the lack of cooperation of even a single landowner could jeopardise the success of a waterbody's protection. In contrast, the microcatchment of a smaller waterbody may be wholly encompassed by one land manager. Additionally, microcatchment deintensification has the potential to provide farmers with locally visible, biodiversity benefits. Such local returns are very important for improving satisfaction and a sense of ownership of the AES measures, as opposed to employing measures for broader scale improvements where results are harder to observe at a site level.

Supplementary to the deintensification of small catchments is the possibility of creating small waterbodies. Ponds and ditches are relatively easy to create and can be located so as to have small, completely unintensive catchments with good water quality, providing the best possible chance of supporting high levels of aquatic biodiversity, whilst involving minimum amounts of land (Williams et al., 2008). This is particularly important given evidence of the time lag between landuse change and improved ecological quality (Harding et al., 1998).

Protection of the aquatic biodiversity of smaller waterbodies through catchment deintensification and the creation of small waterbodies can be undertaken relatively quickly. This means that such initiatives could be employed immediately, providing benefits for aquatic biodiversity across a region much more quickly than could be achieved for larger waterbodies through catchment deintensification, whose complex implementation would take some time to execute.

Clearly, the protection of small waterbodies through catchment deintensification and the creation of ponds cannot deliver the complete protection of aquatic biota within agricultural landscapes (Davies, unpublished data). However, their inclusion amongst measures for aquatic biodiversity protection should provide 'pockets' of high biodiversity in working agricultural landscapes, making aquatic biodiversity protection more effective and more economically efficient.

Conclusion

This study appears the first in the literature to compare both the biodiversity and catchment size of such a range of waterbody types. Small waterbodies, and in particular ponds, were found to support a relatively high proportion of the aquatic biodiversity in the study location, which confirmed the results of the few other studies that have made biodiversity comparisons between a more limited range of waterbody types. Smaller waterbodies were also found to have smaller catchment areas than larger waterbodies, which was not a surprising result, but the implications that arise from it are important.

Calculation of the areas of agricultural land within the catchments of the different waterbody types that would need to be deintensified to provide adequate protection, indicated that, for larger waterbodies, such a method of deintensification would be inappropriate and uneconomic due to the scales that are likely to be involved. In contrast, complete catchment protection would be quite feasible for smaller waterbodies. Given that the smaller waterbodies supported higher levels of aquatic biodiversity, deintensification of their relatively small catchments should afford effective protection from many pollutants, enabling pockets of high biodiversity to exist in a working agricultural landscape. Combined with alternative methods to protect waterbodies with larger catchments, and the creation of strategically located new ponds a landscape matrix should result which incorporates minimally impacted aquatic habitats whilst still being economically and socially productive.

Acknowledgements The authors would like to thank Glen Hart and the Ordnance Survey for provision of MasterMap and Landform Profile data and Geoff Smith and CEH Monks Wood for provision of Land Cover Map 2000 data. We are also grateful to Steven Declerck and two anonymous referees for very useful comments on an earlier draft of this text.

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