Nature based measures reverse catchment biodiversity loss and increase freshwater resilience in an agricultural landscape

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Abstract

This study presents some of the first evidence of the effects of nature based agricultural mitigation measures on freshwater biodiversity at catchment level. We measured alpha (site) and gamma (catchment) richness in all waterbody types (streams, ponds, ditches) present in three upper-catchments in the English lowlands to investigate whether adding (i) ecosystem services measures and (ii) biodiversity protection measures would increase freshwater plant biodiversity. All catchments saw a background decline in macrophytes during the nine-year survey period, with a mean species loss of 1% pa, and a rare species loss of 2% pa. Ponds were a lynchpin habitat with a disproportionate influence on catchment trends. Five years after introducing nature based measures, regression analysis shows that natural colonisation of ecosystem services waterbodies (bunded streams and ditches, runoff ponds, flood storage ponds) largely cancelled-out the background catchment decline in plant richness but, importantly, did not restore the loss of rare plants. The addition of clean water ponds as a biodiversity-only measure brought substantial benefits, increasing catchment richness by 26%, and the number of rare plant species by 181%. Populations of spatially restricted species also increased and other metrics of resilience improved. Adding stream debris-dams had no effect on plant richness and resilience measures. The findings suggest that ecosystem services measures could bring some biodiversity benefits to agricultural catchments. However, the creation of clean-water ponds specifically targeted for biodiversity may hold considerable potential as a tool to help stem, and even reverse, ongoing declines in freshwater plant biodiversity across farming landscapes.

Key words: stream; ditch; clean water pond; constructed wetland; net gain; ecosystem services

1 Introduction

Measures to protect the aquatic environment within farming landscapes currently cost many millions of pounds annually, with expenditure exceeding £470 m per annum in the UK alone (Rayment, 2017). This spend reflects a widespread recognition that agriculture, which makes up about 40% of land cover worldwide and 70% in Britain (Foley *et al.* 2005; Brown *et al*. 2006), plays a major role in modifying, and commonly degrading, freshwater ecosystems and the services that they provide (Moss, 2008; Gordon, *et al.* 2008). In Europe, for example, member states report that nutrient pollution significantly degrades 28% of all surface water bodies classified in the Water Framework Directive (Carvalho et al., 2018).

The mitigation techniques used to protect freshwaters from agriculture-associated impacts are wide-ranging. They include pollutant control measures (e.g. riparian fencing, buffer strips, constructed wetlands, nutrient management plans, minimum tillage), measures to hold back water in catchments, reduce flow, and increase infiltration (e.g. balancing ponds, rural Sustainable Urban Drainage Schemes (SUDS), afforestation), and measures to improve biodiversity and resilience (e.g. debris dams, flow deflectors, river restoration, lake biomanipulation, pond creation and management), (Cuttle et al., 2016, Zhang et al., 2017).

Despite the cost and effort involved in implementing agricultural water protection measures, meta-analyses suggest they are variable in their effectiveness and improvements are often considerably less than anticipated. This is certainly true for freshwater biodiversity where few studies have proven that measures bring significant biodiversity or resilience gains (Bernhardt and Palmer, 2011; Louhi et al., 2011, Harris and Heathwaite, 2012; Robertson et al., 2018). Even where improvements are reported, their relevance is often difficult to gauge. Impressive, statistically significant improvements in metric scores may, for example, prove to be of limited value when put in the wider context of statutory quality targets or minimally impaired reference conditions (Palmer, 2009; Rhodes et al., 2007; Thomas, 2018).

Our limited understanding of the benefits of agricultural measures on freshwater biodiversity is compounded by an almost universal tendency to evaluate measure success based on a very partial element of the catchment: mainly rivers and streams, and in some cases lakes. We now know that small waterbodies including ponds, springs, headwaters and ditches, typically support a high proportion of the freshwater biodiversity within agricultural catchments (Williams et al., 2004; Céréghino, 2007, Davies et al., 2008, Bubíková and Hrivnák, 2018). Ignoring these habitats when assessing the benefits of mitigation measures can lead to a perception bias. The considerable overlap between the biota of standing and running waters (Biggs et al., 2017) means that species gains reported in river networks can be trivial if these taxa are already widely present in other catchment waterbodies. Whilst reliance on data from a subset of catchment waterbodies may miss broader trends that profoundly affect freshwater biodiversity.

In this paper we look at the effect of a introducing a range of nature based water protection measures to higher plant biodiversity and resilience measures within a typical agricultural area of the English Midlands. The work forms part of the Water Friendly Farming initiative: a long-term research and demonstration project to investigate the effectiveness of landscape-wide mitigation measures intended to reduce the impact of rural land use on water, whilst maintaining profitable farming (Biggs et al., 2014).

The study is based on a Before-After-Control-Impact (BACI) design with a three-year baseline and, to date, five years of post-intervention monitoring. The nature based measures we applied included resource protection features intended to slow flows, intercept polluted water and sediment and store flood water (e.g. bunded ditches, interception ponds, flood storage ponds). We also added two simple habitat creation measures specifically intended to bring biodiversity benefits: clean water ponds and stream debris dams. Clean water ponds are off-line waterbodies (not connected to streams or ditches) located in parts of the landscape where, as far as possible, they fill with unpolluted surface-water or groundwater. Recent evidence suggests that these waterbodies can rapidly become species-rich and retain their value for many years (Williams et al., 2008, 2010; Oertli, 2018). Debris dams are widely applied in river restoration as features intended to increase habitat (and therefore by implication, biotic) diversity (Roni et al., 2014).

To assess the impacts these mitigation features had at landscape scale, we undertook site and whole-catchment census studies in all the freshwaters present in the study area, specifically: streams, ponds and ditches. The landscapes have no waterbodies large enough to be described as rivers or lakes (Brown et al., 2006). We chose to use wetland plant attributes as measures of biodiversity and resilience because: (i) census data can be collected relatively rapidly, (ii) temporal and spatial trends in wetland plants have been shown to correlate positively with trends in other biotic groups, particularly aquatic macroinvertebrates, making them broadly representative (Williams et al., 2004; Vad et al., 2017; Zelnik et al., 2018).

2 Study area

The Water Friendly Farming study area lies within three adjacent sub-catchments of the River Welland and the River Soar in Leicestershire, England. Topographically, this is a region of low rolling hills (95-221 m OD) with mixed farming (Table 1). The cultivated land falls into two of the most extensive of Britain's agricultural land classes: Defra Land Class 4 eutrophic tills, and Land Class 6, pre-quaternary clay, which together make up 35% of the cultivated land in Great Britain (Brown et al., 2006). Agriculture in the study area is divided between arable land mainly under oilseed rape and winter wheat with additional field beans or oats, and grassland used to pasture beef cattle and sheep or cut for hay or silage.

3 Experimental set-up

The study is based on a Before-After-Control-Impact (BACI) design with two experimental catchments centred on the Eye (52° 37' 34.4"N 00° 53' 12.4"W) and the Stonton (52° 36' 08.0"N 00° 54' 32.8"W), and one control catchment: the Barkby (52° 38' 48.9"N 00° 54' 28.0"W). Each catchment is around 10 km^2 in area (respectively: 10.6, 9.4, 9.6 km^2). Survey waterbodies were identified using a combination of Ordnance Survey 1:1 250 scale maps, landowner information and field-walks. Three waterbody types were recognised using the definitions given in Williams et al. (2004): streams, ditches and ponds.

Nature based mitigation measures were introduced in 2013 and the early months of 2014, with the exception of debris dams (most introduced in 2015). Two types of measures were added (Table 1):
(i) Ecosystem services measures introduced to both the Eye and Stonton catchments. These were designed to have a multifunctional role including pollution reduction, flood peak attenuation, groundwater recharge and biodiversity protection. They included (a) earth-bunded streams and ditches mainly used to hold-back water and trap sediment and (b) two types of interception ponds: run-off ponds that intercept arable field drains and flood-storage ponds filled by streams or ditches during periods of high water flow.
(ii) biodiversity-focussed measures, which were added to only one of the experimental catchments (Stonton). Two measures were added: clean water ponds and stream debris dams, both specifically designed to increase catchment species richness and resilience.

All measures were left to colonise naturally, with no wetland plants added to the waterbodies or their banks.

The third catchment (Barkby) remained as a control. No mitigation measures were installed here although, as in the Eye and Stonton, this catchment had a normal level of agri-environment scheme protection (e.g. cross compliance buffer strips, Defra 2018).

4 Methods

4.1 Sampling strategy

All catchments were monitored for three years prior to the addition of nature based protection measures (2010-2012). Ecological surveys were not undertaken during 2013 whilst measures were put in place. Post-mitigation monitoring was undertaken annually during the five-year period from 2014 to 2018.

Wetland plant (i.e. aquatic macrophyte) data were collected annually from waterbodies in August. Plant species and their percentage abundance were recorded while walking and wading the margins and shallow water areas of the waterbody. For sites with deeper water, submerged aquatic plants were surveyed using a grapnel thrown from the bank. 'Wetland plants' were defined as the plants listed in Freshwater Habitats Trust (2015) Wetland Plants

Recording Form, which comprises a standard list of ca. 300 water-associated higher plants divided into three categories: submerged, floating-leaved and emergent plants.

Site (alpha) richness data were collected from randomly selected locations that were revisited each year (termed here 'standard samples'). Site selection was stratified by catchment and waterbody type. Twenty standard sample sites were surveyed from each waterbody type in each catchment. This gave a total of 360 sample sites per year prior to the introduction of measures, and around 420 thereafter. River and ditch sections were selected by dividing the network into 100 m lengths and randomly selecting 20 lengths for survey. To ensure that ecological data gathered from different waterbody types could be directly compared, the sampling was area-limited with data from each site collected from a 75 m^2 area of the waterbody based on the method described in Williams et al. (2004). Although this area-based method enabled waterbodies with widely differing dimensions and characteristics to be compared, small waterbodies less than 75 m^2 are by definition, excluded from the survey. To avoid completely omitting smaller habitats, where appropriate, closely adjacent pools were aggregated to give a 75 m^2 total area. This included tree-fall pools in a wooded fen, and small ecosystem services bunded-ditch pools in a series of two or three features. Debris dams were assessed by surveying a 75m^2 area of stream centred on the dam.

Gamma richness data were collected from a census survey of all ponds and ditches in each year. Streams were an extensive habitat type that could not be fully surveyed annually. Our original aim was to use rarefaction curves (Colwell et al. 2012), to calculate gamma richness from a dataset combining standard samples and an additional 10 randomly selected stream samples. In practice the rarefaction curves gave variable and intuitively unlikely results. To investigate this, all streams were fully surveyed in 2018. Comparison of our stream census data and rarefaction-predicted results showed that rarefaction curves over estimated true gamma richness in 2018 by 17%-68% depending on the algorithm used and catchment modelled. Simple summing of the standard and random alpha richness data gave a result that was a close match to the true gamma of our relatively homogeneous streams: varying between zero and one species per catchment below true gamma. In the following analysis we have therefore used the summed stream alpha survey data to represent true gamma for all years.

4.2 Analytical methods

Wetland plant biodiversity was assessed on the basis of species richness and species rarity. Alpha richness was measured as the number of species, or distinctive taxa, recorded. Alpha rarity was the number of regionally or nationally rare plant species recorded in any of the categories listed in Table 2. Gamma richness and rarity were the total number of species and rare species recorded respectively in each waterbody type or catchment area.

Resilience was assessed biologically in terms of alpha and gamma richness and rarity metrics (as above), and in terms of population attributes (Timpane-Padgham et al., 2017), specifically: (i) the number of species with restricted distribution in our catchments, measured as species recorded from only one or two sites, termed 'uniques' and 'doubles' respectively (c.f. Gotelli and Colwell, 2011), (ii) the number of restricted species with small populations, measured as species with a total aerial coverage of <5m^2. Physical metrics of resilience were: (i) total water body area, measured from 1:500 scale map data and ground-truthed in the field, (ii) the proximity of standing waters, measured as a nearest neighbour analysis on ArcGIS Pro 2.3.

Statistical differences between the species richness of waterbody types and catchments were tested using two and three way between-subjects ANOVAs, using square-root transformed data. Non-parametric tests (2-tailed Mann-Whitney U and Friedman tests) were used to analyse plant rarity data where there were a high proportion of zeros. Simple linear regression was used to investigate changes in richness and rarity over time. Analyses were run on IBM SPSS Statistics version 2015.

5 Results

5.1 Baseline wetland plant results for the three catchments

Underlying trends in alpha (site) and gamma (catchment) plant richness and rarity were calculated using data from each annual survey. These data sets excluded new ponds and other waterbodies created or modified after 2013 as part of project measures. The results therefore indicate the background biodiversity trends in the absence of the direct effect of the project's physical habitat creation work.

5.1.1 Background alpha richness and rarity trends

Figure 1 illustrates the underlying trends in alpha richness for each waterbody type in each catchment. A factoral ANOVA to look at the relationship between waterbody type, richness and year showed that the main effect for waterbody type gave an F ratio of $F(2,1368) = 165.653$, $p<0.001$, indicating significant differences in plant richness between the waterbody types. Pairwise comparisons showed a significant differences at $p<0.001$ in all cases, with ponds considerably richer in wetland plant species than other waterbody types (Mean=8.93 species, SE =0.20), followed in turn by streams (M=5.58, SE=0.18) and then ditches (M=4.28 SE=0.13). Ponds were also the only habitat type that consistently supported submerged and floating-leaved plant species.

The main effect for catchment type yielded an F ratio of $F(2,1368) = 20.960$, $p<.001$, indicating significant differences in plant richness between the three catchments. Pairwise comparisons showed that Barkby catchment was significantly richer in wetland plant species than the other two catchments (both $p<.001$), M=7.19 SE=0.21). The Eye catchment (M=6.16 SE=0.19) was marginally, but not statistically significantly, richer than the Stonton catchment (M=5.44 SE=0.18), $p=.070$). There was a significant interaction effect between catchment and waterbody type, ($F(4,1368) = 3.467$, $p=<.01$).

Visual inspection of temporal trends through the survey period (Figure 1) suggests a tendency for mean alpha richness to decline between 2010 and 2018 in most waterbody types and catchments. However, this trend was not provable statistically with no significant main effect for year or for the three-way interaction and between-subject effects for year, waterbody type and catchment.

Rare plant species recorded from the standard survey sites were largely restricted to ponds, which supported a mean of 0.44 rare species per site. The occurrence of rare plants in streams and ditches was an order of magnitude lower (both 0.03 rare species per site). The high proportion of zero values in the dataset precludes tests of statistical significance. The number of rare plant species did not differ significantly between any of the three catchments (Mann-Whitney U test). There was no evidence of temporal trends in the rare species data for any catchment (Friedman test of differences among repeated measures).

5.1.2 Gamma richness and rarity trends

The underlying gamma richness trends for all waterbody types and catchments in the absence of measures are shown in Figure 2. In total 106 wetland plant species were recorded from census surveys of the three catchments during this time, with a mean of 92.8 species (range 89-99) in any one year. The Barkby control catchment supported the greatest number of species per year (78.9 species, 85% of total gamma for all three catchments), followed by the Stonton (61.3 species, 66% of total gamma) and Eye (54.0 species, 59% of total gamma).

Gamma richness differences between the waterbody-types broadly concurred with the alpha richness data. Ponds were much the richest waterbody type in all three catchments in all years, supporting an average of 85.5 species per year. Streams and ditches each supported around half this (43.5 and 38.9 species, respectively).

Temporal trends in gamma richness provide evidence of species loss over the nine-year survey period (Figure 2). In the absence of measures, a simple linear regression showed that gamma richness across all waterbody types and catchments combined declined by 10% during the survey period: ($F(1,6)$=13.785, p<.01) with an adjusted R^2 of .646. This represents a loss of 1.1 wetland plant species per annum across the combined area of the three catchments.

Within individual waterbody types, both streams and ditches saw significant declines in richness through the survey period. The regression equation for streams indicates a decline of 1.2 species per annum ($F(1,6)$= 43.774, p<.001) with an adjusted R^2 of .859. The rate of loss in ditches was 0.7 species per annum ($F(1,6)$= 9.755, p<.05) with an adjusted R^2 of .556. Pond richness also showed a tendency towards decline, however declines in the Barkby and Eye catchment ponds were counter-balanced by increased richness in the Stonton Ponds (see below), and the combined regression equation was not statistically significant ($F(1,6)$= 1.545, p=.26) with an adjusted R^2 of .072.

Of the three catchments, the Barkby catchment, which was initially the most species-rich, saw the greatest decline in plant species (14.0%), $F(1,6)$= 29.742, p<.01) with an adjusted R^2 of .804 representing a loss of 1.4 species per year. The Eye catchment declined by 10% $F(1,6)$= 7.987, p<.05) with an adjusted R^2 of .500, 0.6 species per year. Declines in the Stonton catchment were smaller (2%) and not statistically significant $F(1,6)$= 0.340, p=.581) with an adjusted R^2 of -.104.

Amongst individual waterbody types and catchments, there was just one incidence where species richness tended to increased rather than decrease during the survey. This was pond gamma richness in the Stonton catchment where simple regression indicates an increase of 5% over the nine-year period (0.3 species pa). This increase is too small to be significant ($F(1,6)$= 1.366, p=.287) with an adjusted R^2 of .050. However, the Stonton ponds remain anomalous as the only catchment waterbody type that showed a net gain in species between the first and last year of the survey. Examination of the data suggests that at least part of this increase was due to an unexpected indirect effect of the project's habitat creation after 2014: with new-to-the-catchment species, like the submerged aquatic *Potamogeton pusillus* and marginal plant *Equisetum palustre,* that rapidly colonised the new clean-water ponds, subsequently moving out to colonise adjacent pre-existing ponds.

In total, 18 plant species that were either nationally or regionally rare were recorded from the baseline waterbodies in the three catchments across all years; 16% of the total flora. Ponds supported by far the greatest proportion of these rare species (89% across all catchments and survey years). Streams and ditches each supported 17% of the rare species pool.

All catchments showed a tendency towards a background loss of rare plant species during the survey period (Figure 3). Simple regression of total gamma rarity for all waterbodies and catchments combined gives a decline of 22% (0.4 species pa), ($F(1,6)$= 8.088, p<.05) with an adjusted R^2 of .503.

Of the three catchments, declines were greatest in the Barkby where there was a loss of 0.5 rare species per annum (loss of 34% of the Barkby's rare plant species during the sampling period), ($F(1,6)$= 13.224, p<.05) with an adjusted R^2 of .636. Rare species loss was also high in the Eye catchment: 0.3 species per annum (38%), ($F(1,6)$= 10.456, p<.05) with an adjusted R^2 of .575. The Stonton catchment showed a small net decline in rare species, but the regression relationship is not significant (F(1,6)= 0.576, *p*=.476) with an adjusted R^2 of -.064. In all catchments, loss of rare plant species was predominantly (63%-75%) due to the loss of submerged aquatic plant species from the catchments' ponds.

5.1.3 Unique and restricted species

Combining data from all catchments and all years shows that close to half (48%) of the plant species recorded from baseline waterbodies were found in only one of the three waterbody types. Of these species, 83% were restricted to ponds, 13% to streams and 4% to ditches. This pattern was more striking for rare species where 89% of the baseline plant species were restricted to a single waterbody type, and 88% of these were unique to ponds. Two species were unique to other waterbody types, but both of these were lost early on in the survey due to adverse management practices.

Many plant species had a highly localised distribution and small populations. In the final year of the survey 34% of taxa were found only as uniques or doubles (i.e. recorded from one or two sites respectively) across the three catchments. Of these localised taxa, 90% were found in ponds, and 83% only in ponds; 10% were recorded only in streams and 3% only in ditches. Most uniques and doubles also occurred at low abundance, with 70% occupying a total area of less than 5m^2 across all catchments. The majority (86%) of these were pond-only species. Combining these data shows that 23% of all remaining plant species across the three catchments were both present at very few sites and occurred in low abundance within those sites. 86% of these highly restricted plant species were unique to ponds.

5.2 Effect of adding nature based measures

5.2.1 Physical effect of adding measures

Two main types of nature-based measure were introduced to the project area in 2013. Multifunctional ecosystem services measures (interception ponds, bunded streams and ditches) were added to both the Eye and Stonton catchments increasing the area of freshwater habitat by an average of 0.2 ha per catchment (Table 1). Biodiversity-only measures: (clean water ponds, debris dams), which were introduced to the Stonton catchment alone, added an extra 0.24 ha to this catchment. Together, the nature based measures increased the area of standing waters present in the Eye and Stonton by an average of 33%, and the total area of all freshwater habitats in these catchments by an average of 9%. Nearest neighbour analysis for standing waters showed that, in the Eye catchment, the effect of adding ecosystem services measures was to reduce the between-waterbody distance of standing waters by 29% from an average of 336 m to 240 m. In the Stonton, where clean water ponds were also added, there was a 64% reduction from 255 m to 92 m between waterbodies.

5.2.2 Alpha richness and rarity of measures waterbodies

In the final survey year, five years after their creation, the mean alpha richness of the new clean water ponds was significantly greater than other nature based waterbody types created (p <.001 for all analyses, Figure 4a). Plant richness associated with debris dams was universally low, and significantly less than for other measures (Table 3). Amongst the ecosytem services measures, interception ponds were significantly richer than bunded ditches and streams (p <.01).

The alpha richness of the new clean water ponds was also significantly greater than the richness of all pre-existing waterbody types in the two experimental catchments (Table 3). The richness of interception ponds was similar to pre-existing ponds. Bunded ditches and streams had a similar mean richness to pre-existing streams, and were marginally richer than pre-existing ditches (p=.049), (Figure 4).

Stream richness adjacent to debris dams was significantly lower than was typical of pre-existing streams (p<.05), and was more similar to the richness of ditches. It is likely that this is because debris dams were typically added to smaller streams, most of which were also heavily shaded. This is supported by comparison between debris dams sections and the nearest unaffected stream lengths which shows that their average richness was almost identical (debris dams 2.6 species; adjacent streams 2.7 species).

Of the four main types of nature based measures, only clean water ponds and interception ponds supported rare species. At five years old, the mean alpha rarity of clean water ponds exceeded that of other nature based and pre-existing waterbody types and was around double that of pre-existing ponds (Figure 4c). The high proportion of zeros values in the data set precludes tests of statistical significance.

5.2.3 Gamma richness and rarity of measures waterbodies

Amongst the introduced measures, clean water ponds supported the greatest gamma richness, followed by interception ponds, bunded watercourses and debris dams (Figure 4b). However, the gamma richness of both pre-existing ponds and streams exceeded the total richness of any of the measures. Gamma rarity showed a different trend (Figure 4d) with the new clean water ponds supporting the greatest number of rare species, followed by the pre-existing ponds. Interception ponds supported few rare species, and they were absent from bunded watercourses and debris dams.

5.2.4 Catchment scale effects from introducing measures

The addition of nature based measures had notable catchment-scale effects in both the Eye and Stonton catchments (Figures 5, 6).

In the last year of the study (2018), when the measures were five years old, new-to-the-catchment plant species in ecosystem services waterbodies increased the total wetland plant richness of the Eye catchment by 14%, from 50 species in all pre-existing waterbodies, to 57 species including the ecosystem services measures (Figure 5a). Simple regression over the survey period, shows that adding ecosystem services measures to the Eye catchment converted a statistically significant loss of species in pre-existing waterbodies to a small, non-significant gain ($F(1,6)= 1.442$, $p=.275$) with an adjusted R^2 of .059. In the Stonton catchment, ecosystem services measures increased species richness by 7%, (Figure 5b). This changed the small non-significant downward trend in richness in pre-existing waterbodies to a small non-significant upward trend ($F(1,6)= 1.883$, $p=.219$) with an adjusted R^2 of .112.

The clean water ponds contributed more substantially to the Stonton catchment. In 2018, the five-year old clean water ponds added 13 new wetland plant species; a 22% increase from 60 species in all pre-existing waterbodies, to 73 species with the clean water ponds (Figure 5c). Simple linear regression for the survey period as a whole, shows that this changed the small non-significant trend towards loss of species in the Stonton catchment to a significant 26% gain ($F(1,6)= 12.318$, $p<.05$) with an adjusted R^2 of .618.

The total number of species contributed by all nature based measures in the Stonton catchment was similar to the number added by the clean water ponds alone (Figure 5d). This was because few plant species were unique to the ecosystem services ponds and debris dams added no new plant species to the Stonton catchment. The combined effect of adding all nature based measures was a statistically significant increase of 27% based on simple linear regression of data for all years ($F(1,6)= 15.116$, $p<.01$) with an adjusted R^2 of .668.

Both the ecosystem services and clean water ponds supported rare species that were new to, or had recently become extinct from, their catchments (Figure 6). However, in the new ecosystem services waterbodies rare species were transitory colonisers and did not persist after the features were 1-3 years old (Figures 6a,b). The clean water ponds made a substantial and lasting contribution to catchment rarity. In 2018, the final year of the survey, pre-existing waterbodies in the Stonton catchment supported only four rare species. Adding the clean water ponds tripled this to 12 species. Linear regression for the survey period as a whole, shows that their addition changed the small non-significant trend towards rare species loss of species in the Stonton catchment to a significant gain of 181% ($F(1,6)= 35.943$, $p=.001$) with an adjusted R^2 of .833 (Figure 6c). New rare species that colonised the clean water ponds included the Near Threatened *Triglochin palustris*, together with species such as *Isolepis setacea*, *Juncus subnodulosus* and *Hippuris vulgaris* that are both rare in the region and increasingly uncommon in lowland England.

5.2.5 The effect of adding nature based measures on other resilience attributes

On average, 18.5% of species that had been recorded in the pre-existing Eye and Stonton waterbodies (2010-2017), were no longer recorded in these catchments in 2018. Of these ‘lost’ species 15% were present in the Ecosystem Services waterbodies. In the Stonton catchment where clean water ponds were also created, 31% of ‘lost’ species were present in these ponds. No ‘lost’ species were retained as a result of creating the debris dams.

In the final year of the study both types of nature based measures created habitats for populations of plants that were otherwise highly restricted in their catchments. In total, the clean water ponds provided 34 new sites for six (22%) of the plant species that were otherwise present only as uniques or doubles in the Stonton catchment. Ecosystem services ponds added an average of seven new sites for 3.5 species (12%) otherwise present as uniques or doubles in the Eye and Stonton catchment.

6 Discussion

6.1 Trends in catchment aquatic biodiversity

This study is amongst the first to look at catchment-level temporal trends in wetland plant richness across the full range of waterbody types present in a typical agricultural landscape. The results showed systematic evidence of decline over the nine-year survey period, with an annual loss of 1% of plant species, and 2% loss of rare species across the 30 km^2 survey area as a whole. The limited availability of comparable data makes it difficult to know how typical these trends are of wetland plant assemblages in other lowland agricultural catchments. The main exception is Gołdyn (2010) who undertook a 30 year re-survey of plant richness across a range of agricultural waterbodies in West Poland. She found an increase in gamma richness between 1976 and 2007, although this was mainly due to colonisation by alien and ruderal species, and the number of rare and threatened plants declined. Of the available data for specific waterbody types, the most directly comparable come from the UK Countryside Survey, which looked at trends in headwater stream, ditch and pond plant assemblages across Britain over recent decades. For ponds at least, Countryside Survey results broadly parallel our study findings, showing that in the English lowlands, pond alpha plant richness declined by around 20% (1.8% pa) between 1996 and 2007 (Williams et al 2010). Data from a separate study of ponds in northern England over the same time period show a similar loss of gamma diversity when based on compatible wetland plant species lists (recalculated from Hassall et al., 2012). For English headwater streams, Countryside Survey data showed the reverse trend, with stream plant alpha richness increasing significantly between 1998 and 2007 (Dunbar et al., 2010). This contrasts both with our findings, and with the majority of data from studies of larger European watercourses, which typically show declining alpha and/or gamma plant richness over recent decades (Riis and Sand-Jensen, 2001; Gerhard et al., 2016; Schütz et al., 2008;

Steffen, 2013). Comparable data for headwater ditches are hard to come by, with almost all longitudinal studies focused on the wetland flora of more permanent ditches on floodplains, coastal wetlands or semi-natural wetlands, rather than the small, and often seasonal ditches typical of England's agricultural countryside (Best, 1995; Drake et al., 2010; Whatley, 2013). The main exception is, again, the UK Countryside Survey, which in contrast to our findings, suggests that mean wetland alpha plant richness in headwater ditches increased in the decade before our study was undertaken (Dunbar et al., 2010). Overall, therefore, the results for individual waterbody types provide evidence of a systematic continued loss of plant diversity from agricultural ponds in England's lowlands, but suggest that trends in stream and ditch flora may be more temporally or spatially variable in agricultural headwaters.

6.2 Future losses

Looking forwards, it is clear that a high proportion of the wetland flora in our survey area remains vulnerable to further loss. During the 2010-2018 survey period we directly observed the extinction of species from all catchments and waterbody types through habitat change and destruction including culverting of springs, afforestation of fens and cessation of grazing along waterbody margins. Given that a third (34%) of our catchments' remaining plant species were restricted to two or fewer sites, and almost a quarter (23%) of these species occurred at exceptionally low abundance, the risk of further loss as a direct result of habitat impacts seems considerable. An additional threat comes from the extent of habitat isolation inherent in many agricultural landscapes (Bosiacka et al., 2008). Both theoretical and empirical studies of extinction debt (Tilman et al., 1994; Loehle and Li, 1996) have shown that fragmented habitats with very small populations have high species extinction rates (Halley et al., 2016). Indeed the risk may be especially high for wetland plants located in spatially discrete wetlands, including ponds (Deane et al., 2017; Deane and He, 2018). This suggests an particular vulnerability in our catchments where the majority (over 80%) of plant species with highly restricted distribution occur only in ponds. Ponds also supported the greatest number of rare species in our catchments and, particularly in the Barkby, suffered considerable loss of these taxa during the nine-year study. Extinction debt is held to be especially likely for rare species (Dullinger et al., 2013), which may help to explain why loss of rare species from our catchments has been so high and indicates that the demise of the rare taxa remaining in our catchments may continue.

6.3 Biodiversity gain from nature based measures

The addition of nature based measures brought almost immediate biodiversity gains to the two catchments in which they were introduced. The creation of clean water ponds, in particular, more than compensated for recent losses and, after five years, had increased whole catchment richness by aproximately a quarter and tripled the number of rare plant species present. The rapid colonization rates observed in new nature based ponds is not entirely unexpected. Authors from Darwin onwards have noted the mobility of many freshwater taxa (Darwin, 1859; Talling, 1951; Soons, 2006) whilst other studies have shown that ponds can develop rich plant assemblages within five to six years (Williams, 2010; Mitsch, 2012). New clean water ponds have sometimes been shown to be richer than pre-existing ponds within two to three years (Parikh and Gale, 1998; Williams et al., 2010).

The unusually high proportion of rare species that appeared in the catchment's clean water ponds is more surprising, but still not unprecedented. Williams et al. (1998) showed that 6-12 year-old ponds supported more uncommon plant species than older ponds in lowland Britain, whilst Fleury and Perrin (2004), found that populations of threatened temporary pond plants were greatest in the first 2–3 years after ponds were created. Beyond this, there are many anecdotal examples of nationally rare plant species appearing in newly created ponds, particularly when they are located in semi-natural habitats (e.g. Barnes, 1983; Kennison, 1986; Erskine et al., 2018).

The reason that rare plant species show a propensity to colonise new ponds is not completely clear. Rare species are often held to be poor competitors (Buchele et al.,1992; Shimada and Ishihama, 2000; Lloyd et al., 2002; Cacho and Strauss, 2014) and the bare substrates of new ponds may provide a competitor-free zone in the first few years after creation. Equally, recently excavated waterbodies tend to have relatively nutrient-poor substrates which may directly benefit groups including charophytes that have been shown to thrive below relatively low nutrient thresholds (Lambert and Davy, 2011). Alternatively, remnants of soil left after pond creation may simply have exposed previously buried seed banks, allowing rare plants that are no longer present in the standing flora to germinate (Nishihiro et al., 2005). In our study, it seems likely that more than one factor was at play. A proportion of the nationally and regionally rare species that we recorded (e.g. *Triglochin palustris, Juncus subnodulosus, Isolepis setacea*) appeared only in ponds created in an area of secondary woodland partly planted on a former fen. *Triglochin palustris*, at least, had previously been recorded from the fen, and although our pre-excavation surveys did not find it in the area where the ponds were created, there seems a high likelihood that this species germinated from a pre-existing seed bank. Other taxa, including *Hippuris vulgaris* and *Chara* species, appeared in new ponds that were located in isolated dry ground areas where colonising species can only have arrived though wind or bird transported propagules (Soons, 2006; Merel et al., 2008). Here, their successful colonisation presumably reflects the new opportunities provided by the ponds' bare substrates.

6.4 Resilience

It can be argued that both the clean water ponds and ecosystem services waterbodies, added resilience to the catchments in our study, although the clean water ponds made a substantially greater contribution. Resilience has been measured in many ways (Cumming et al., 2005; Timpane-Padgham et al., 2017; Beller et al., 2018). Species richness is a common metric, with both alpha and gamma richness linked to increased resource use and community potential to resist, or recover from, environmental stresses (Duffy 2009, Seavy et al., 2009; Mariotte et al., 2013; Hisano et al., 2018). As noted above, both the ecosystem services waterbodies, and particularly the clean water ponds, increased catchment richness. Population size, another correlate of resilience (Stubbington, 2012; Hill, 2016; Aalto, 2019), also increased as a result of adding nature based measures: creating new sites for species that, without them, would be present only as uniques or doubles.

The physical creation of the new waterbodies themselves added a range of catchment-scale attributes held to confer resilience. Total area has consistently proven to be a critical metric linked to long-term habitat resilience (Seavy et al., 2009; Angelini and Silliman, 2012; Ziegleret al., 2017; Fahrig, L., 2019). In our experimental catchments, adding nature based measures increased the area of standing waters by a third, with a 9% gain if all waterbody types are included. Habitat proximity, another widely cited metric with links to dispersal potential and connectivity among populations and communities also increased (Caissie, 2006; Thiere et al., 2009; Chester and Robson, 2011; Ruhí et al., 2016: Schofield et al., 2018), particularly in the Stonton catchment where addition of both ecosystem services waterbodies and new clean water ponds reduced nearest neighbour distances by 64%. Beyond this, it is possible that short-term resilience within the experimental catchments has improved as a result of greater habitat heterogeneity (Pope et al., 2000; Folke et al., 2004; Krosby et al., 2010; Brauns et al., 2011), particularly through the provision of new unshaded, early succession habitats which were previously uncommon in our catchments. In the Stonton catchment, the addition of clean water ponds may also provide a longer-term resilience benefit through the provision of habitats of higher than average quality (Niemi et al.,1990; Brauns et al., 2011).

Our data provide circumstantial evidence that factors linked to the greater spatial connectivity provided by the nature based measures may have, indeed, conferred greater catchment resilience. Of the three catchments, the Stonton, which received the greatest number and widest range of nature based measures, also saw the lowest levels of background species loss. Similarly, the only pre-existing catchment waterbodies to increase in mean richness during the survey period were the Stonton ponds; the habitat type which would be expected to benefit most from the increased connectivity and dispersal opportunities created by adding new ponds in their vicinity. Looking in detail at species trends provides evidence to support this. For example, the submerged aquatic *Potamogeton pusillus*, was not present in the Stonton catchment before the nature-based measures were introduced. However, it rapidly colonised a number of the new clean water ponds, and was subsequently recorded in first one, and then a second pre-existing pond, around 1 km distant, providing a strong indication that the new ponds acted as a stepping stone habitat that ultimately led to greater alpha and gamma richness in the Stonton's pre-existing waterbodies.

6.5 Effectiveness of ecosystem services waterbodies

There is often an assumption that nature based measures will be 'good for wildlife', and that ecosystem services features, such as SUDs waterbodies, agricultural bunded ditches and interception ponds will inevitably provide multi-functional benefits that include biodiversity gain (Burgess-Gamble et al., 2017). In practice, studies to test these assumptions have shown mixed results depending on the type of measures introduced and the biotic group used to derive metrics (Hansson et al., 2005; Thiere et al., 2009; Wiegleb et al., 2017). The results from our plant-based study provides some support for the value of ecosystem services waterbodies, particularly for reversing the impact of short term losses in catchment plant richness, and increasing the number of populations of some unique and replicate species. However, ecosystem services waterbodies did not restore the loss of rare plant species from our catchments and fell far short of clean water ponds in the extent to which they increased catchment resilience. These findings tally with the majority of other plant-based studies of waterbodies and wetlands which have shown that ecosystem service features tend to support more homogeneous communities, and fewer high quality or rare plant species, than their semi-natural equivalents (Balcombe et al., 2005; Brooks et al., 2005; Aronson and Galatowitsch, 2008; Robertson et al., 2018; Price et al., 2019). Taken together, these findings suggest a need for some caution, or at least realism, when promoting the value of ecosystem services waterbodies for plant biodiversity, particularly where waterbodies are on-line and inevitably compromised by pollutant inputs.

6.6 Will the gains persist?

Whether the biodiversity gains contributed by the nature-based measures will persist in the longer-term remains an open question. Evidence from other new clean water ponds shows that this waterbody type can retain high biodiversity for many decades (Williams et al., 2007, Williams et al., 2010). However, in the majority of cases these ponds were created in semi-natural landscapes and further monitoring is required to determine whether the clean water ponds in our agricultural landscapes can maintain their disproportionate contribution to catchment biodiversity in the longer term. The future biodiversity benefits from the ecosystem services ponds seems more doubtful. Evidence suggests that new ponds in highly impacted landscapes tend to reach their greatest richness within the first five years after creation and then decline. This is almost certainly because they degrade once they begin to fill with polluted sediment and water (Williams, 2007 Robertson et al., 2018). For on-line ecosystem service features like ours, where the waterbody's main function is to intercept polluted sediment and water, short to medium term declines in biodiversity value seem likely. Some decline is already evident from our data: a number of ecosystem services waterbodies supported uncommon plants (*Potamogeton* and *Chara* species) in the first few years after creation which were rapidly lost as these features began to fill with sediment and became more algal dominated. It is possible that regularly desilting ecosystem services waterbodies may be able to remove sediment and pollutants and return them to an earlier, and botanically richer successional stage, but this possibility is yet to be tested. Further surveillance data are essential to measure both the extent of further losses, and the potential for de-silting to reset the clock.

7 Implications

7.1 The value of gamma richness data

Alpha richness measures remain a mainstay of ecological research. However, over the last few decades there has been an increasing trend towards collection of gamma data, particularly for freshwater habitats (Bubíková and Hrivnák, 2018). Most studies measure gamma richness as summed alpha values rather than adopting what can sometimes be an impossibly time consuming census approach. In this study we used wetland plants which are a relatively quick and easy group to survey in order to undertake census surveys of waterbodies, and found that this provided advantages for assessing catchment-scale change. For example, alpha data could not provide evidence that the observed species losses were real because between-year variability in site richness confounded statistical significance tests. Gamma census data, which reveal the real world, not only enabled us to measure losses and gains over the relatively short time-scale of the project, but provided confidence in measuring resilience attributes such as the number of restricted species, area of occupancy, and species movement between waterbodies. We suggest that census data could be useful for other freshwater studies, particularly to identify trends over short timescales where changes are likely to be subtle. Census studies that investigate the net change across a wider range of landscapes (agricultural, urban or semi-natural) would be particularly welcome in order to place the results recorded here in a wider context.

7.2 Ponds as catchment controlling habitats

Our findings suggest that ponds were a lynchpin habitat in the agricultural catchments that we studied. By a considerable margin, ponds supported the greatest number of freshwater plant species, the most uncommon species and the highest proportion of unique taxa, with 40% of species only found in ponds. Loss of pond species, particularly submerged aquatic taxa, had a disproportionately high impact on catchment richness and rarity trends.

Recognition of the importance of ponds for supporting catchment biodiversity has been growing for almost two decades (reviewed in Biggs et al., 2017 and Hill et al., 2018). Calls for greater representation of ponds within national and international policy have been growing apace (Williams et al., 2004; Kristensen and Globevnik, 2014; Sayer, 2014; Hassall, 2014; Biggs et al., 2017). Yet there is still no requirement to monitor ponds as part of water quality assessment in Europe, the US or other countries where river monitoring is mandatory (Hill et al., 2018). Equally lacking are water and nature conservation policies that support pond protection at site or catchment level (e.g. River Basin Management Plans) despite increasing evidence that, *en masse*, small habitats like ponds are critical for maintaining landscape scale biodiversity (Hill et al., 2018; Grasel et al., 2018; Fahrig, 2019). The evidence from our study adds further weight to the argument that ponds need to be specifically included in policy and legislation if we are to reduce freshwater biodiversity loss and stand a chance of making sustainable catchment management a reality.

7.3 Good news for biodiversity protection

Amidst the gloomy reality of global declines in freshwater habitats and species, the results from this study provide evidence that proactive habitat creation measures can make a positive difference to agricultural catchment-scale biodiversity over comparatively short time scales. Our results provide the first demonstration of a whole-landscape increase in freshwater biodiversity as a result of agricultural land management measures. The findings emphasise the potential for clean water pond creation to bring significant benefits to catchment biodiversity and resilience in agricultural landscapes.

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Table 1. Physical characteristics of the three survey catchments and nature based features added by the project. The table shows the length and number of waterbodies, with their total area in parentheses

Catchment				**Barkby**	**Eye**	**Stonton**
Treatment				Control	Experimental: ecosystem-services	Experimental: ecosystem-services & biodiversity
Catchment area (km^2)				9.6	10.6	9.4
Landuse		Arable		37%	45%	44%
		Grass		52%	42%	41%
		Woodland		7%	9%	10%
		Settlements & minor landuses		4%	4%	5%
Pre-existing waterbodies		Streams: km length (area ha)		10.70 (1.53)	13.45 (1.91)	9.55 (1.34)
		Ditches: km length (area ha)		2.81 (0.28)	4.27 (0.36)	2.81 (0.36)
		Ponds: number (area ha)		25 (1.12)	20 (0.46)	20 (2.40)
Nature based measures	Ecosystem-services measures	Earth-bunded ditches and streams: number (area ha)		0	9 (0.05)	12 (0.08)
		Interception ponds: number (area ha)	Runoff ponds	0	3 (0.04)	5 (0.04)
			Flood storage ponds	0	4 (0.13)	5 (0.08)
	Biodiversity-focused measures	Clean water ponds: number (area ha)		0	0	20 (0.24)
		Debris-dams: number		0	0	20

Table 2. Definition of rare species used in the study

Regionally rare species: species recorded from fewer than 15% of 1 km grid squares in a 100 km^2 grid square centred on the project area (BSBI, 2019)
Nationally Scarce: recorded from 16 to 100 10x10 km grid squares in Britain (JNCC, 2019)
Red Listed at England or UK level based on the IUCN categories: Near Threatened, Vulnerable, Endangered or Critically Endangered (Stroh et al., 2014)

Table 3. Mann-whitney U-test results comparing the alpha plant richness of nature based measures and pre-existing waterbodies. Statistically significant results in bold

		Bunded ditches & streams	Interception ponds	Clean water ponds	Pre-existing ponds	Pre-existing streams	Pre-existing ditches
Debris dams	U	91.500	34.500	0.000	86.000	247.000	305.000
	Z	-3.124	-4.281	-5.493	-4.944	-2.415	-1.505
	p	**.002**	**<.001**	**<.001**	**<.001**	**.016**	.132
Bunded ditches & streams	U		89.500	5.500	227.000	394.500	291.500
	Z		-2.822	-5.427	-2.946	-0.390	-1.970
	p		**.004**	**<.001**	**.003**	.697	**.049**
Interception ponds	U			32.000	358.000	198.000	118.500
	Z			-4.438	-0.034	-2.735	-4.082
	p			**<.001**	.973	**.006**	**<.001**
Clean water ponds	U				85.500	49.000	7.500
	Z				-5.091	-5.645	-6.282
	p				**<.001**	**<.001**	**<.001**

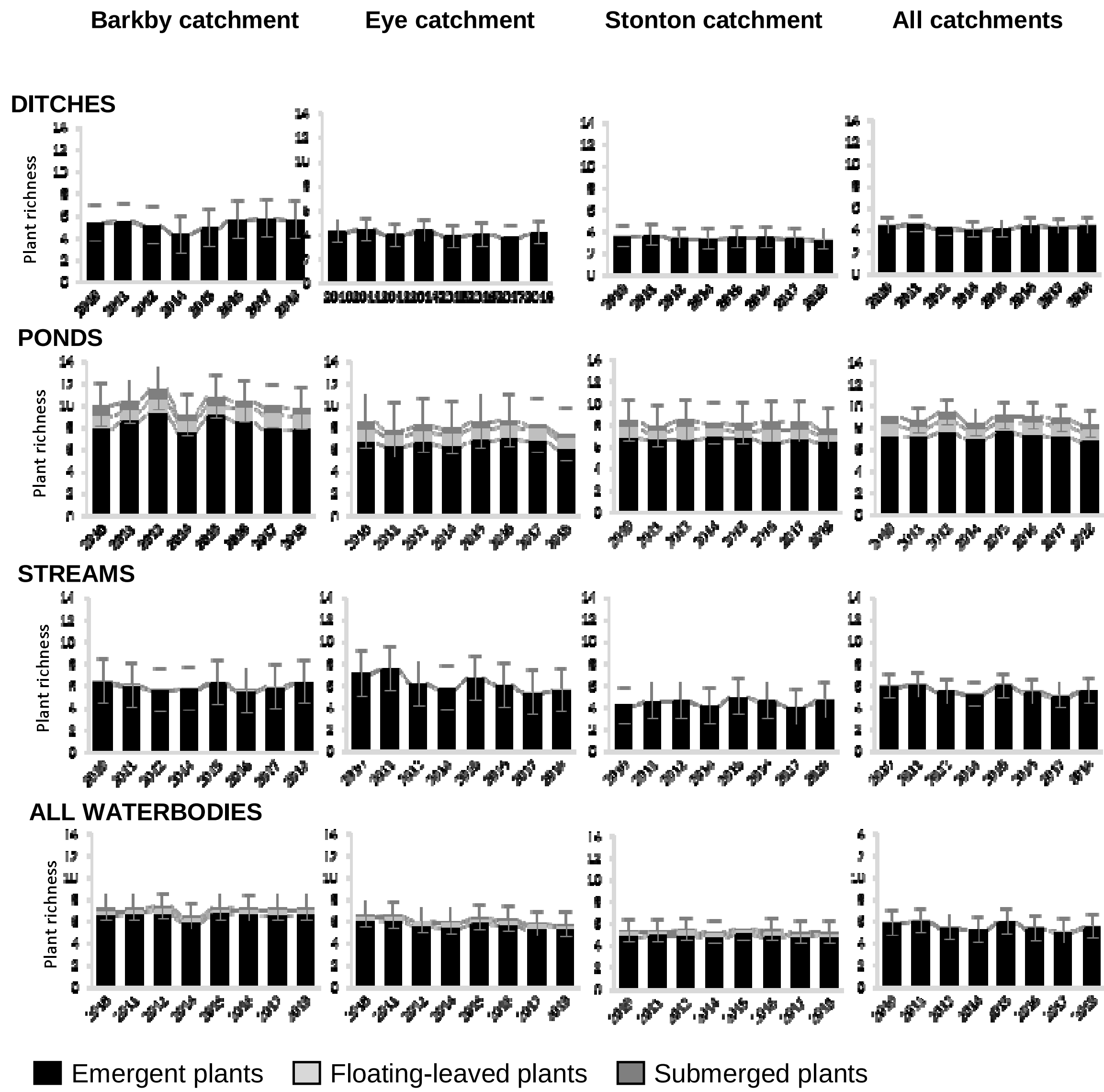

Figure 1. Alpha richness in all waterbody types and catchments: showing the mean number of wetland plant species recorded from annual surveys of 75m^2 plots from 20 ditches, ponds and streams in each of three catchments. No data were collected in 2013 when measures were being introduced. The graphs do *not* include new waterbodies or features added after 2014, and hence show underlying trends in the absence of nature-based measures. Error bars show 95% CLs for total mean richness.

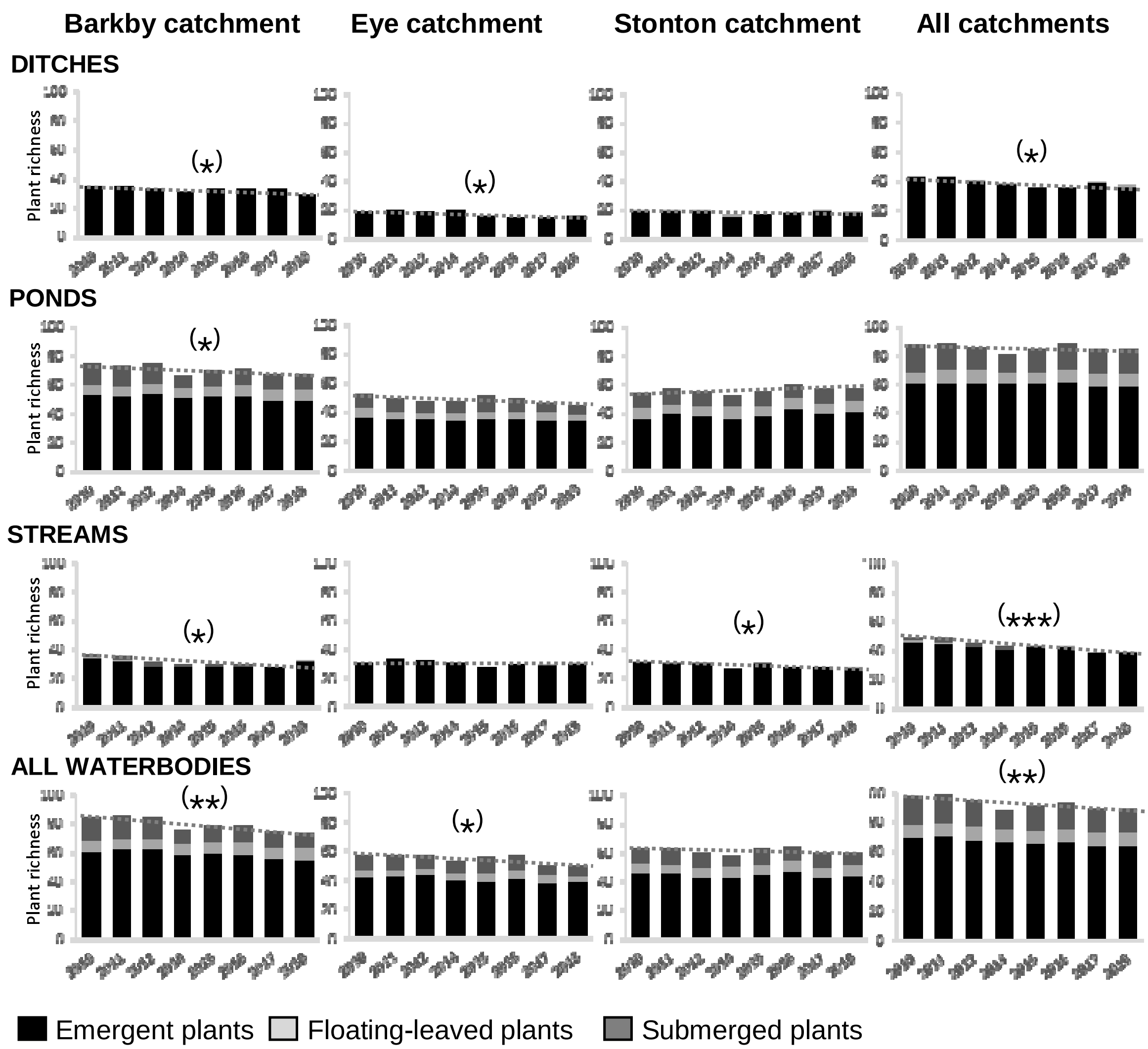

Figure 2. Plant gamma richness for all ditches, ponds and streams in each of three catchments, shown with a line of best fit. No data were collected in 2013 when measures were being implemented. The graphs do *not* include new waterbodies or features added after 2014, and hence show underlying trends in the absence of nature based measures. Dotted lines show the simple linear regression for total plant richness in each waterbody and catchment. Asterisks denote the statistical significance of the regression equation: p-value <0.05 (**), < 0.01 (**), < 0.001 (***).

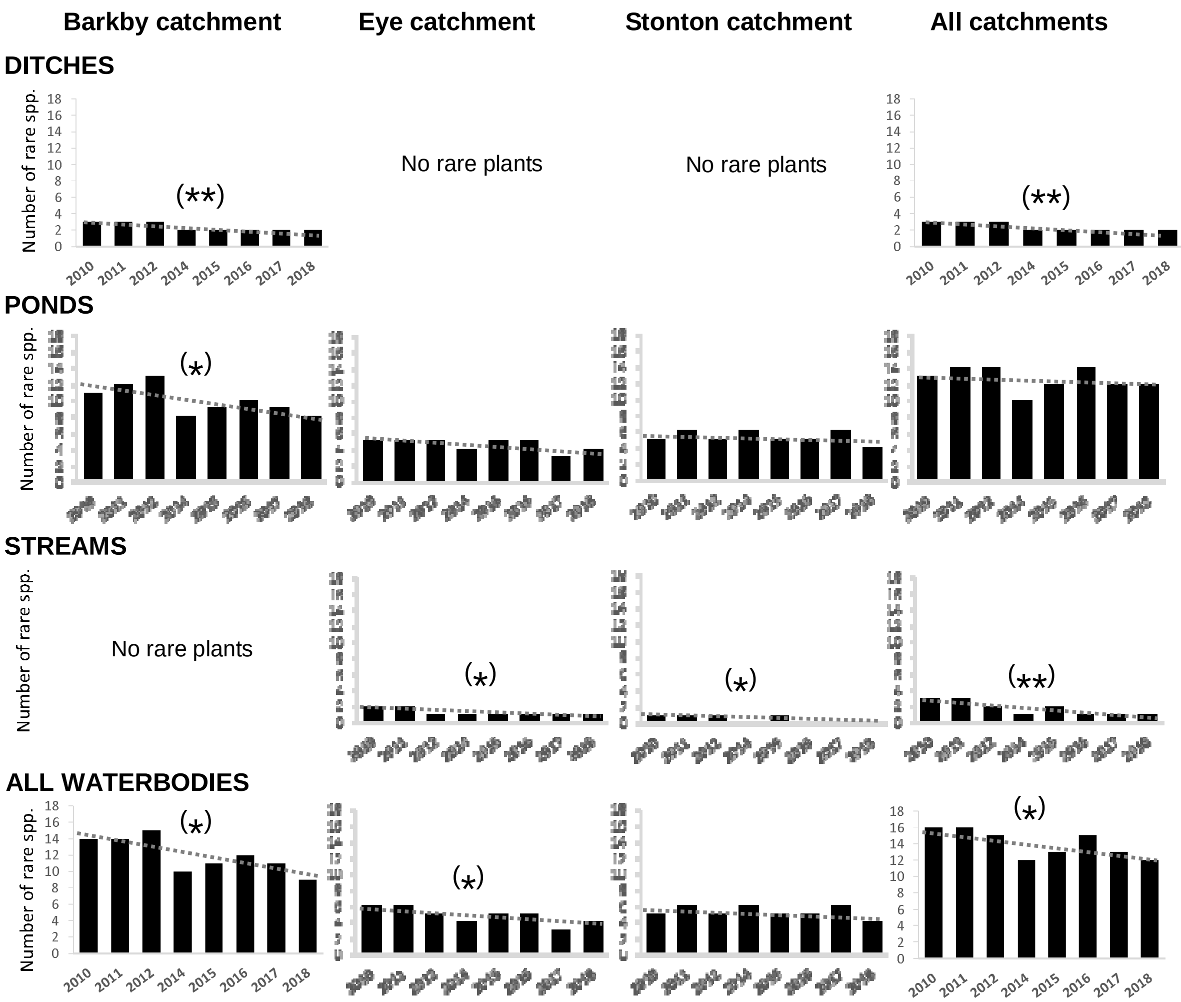

Figure 3. Plant gamma rarity for all ditches, ponds and streams in each of three catchments, shown with a line of best fit. No data were collected in 2013 when measures were being implemented. The graphs do *not* include new waterbodies or features added after 2014, and hence show underlying trends in the absence of nature based measures. Dotted lines show the simple linear regression for total plant richness in each waterbody and catchment. Asterisks denote the statistical significance of the regression equation: p-value <0.05 (**), < 0.01 (**), < 0.001 (***).

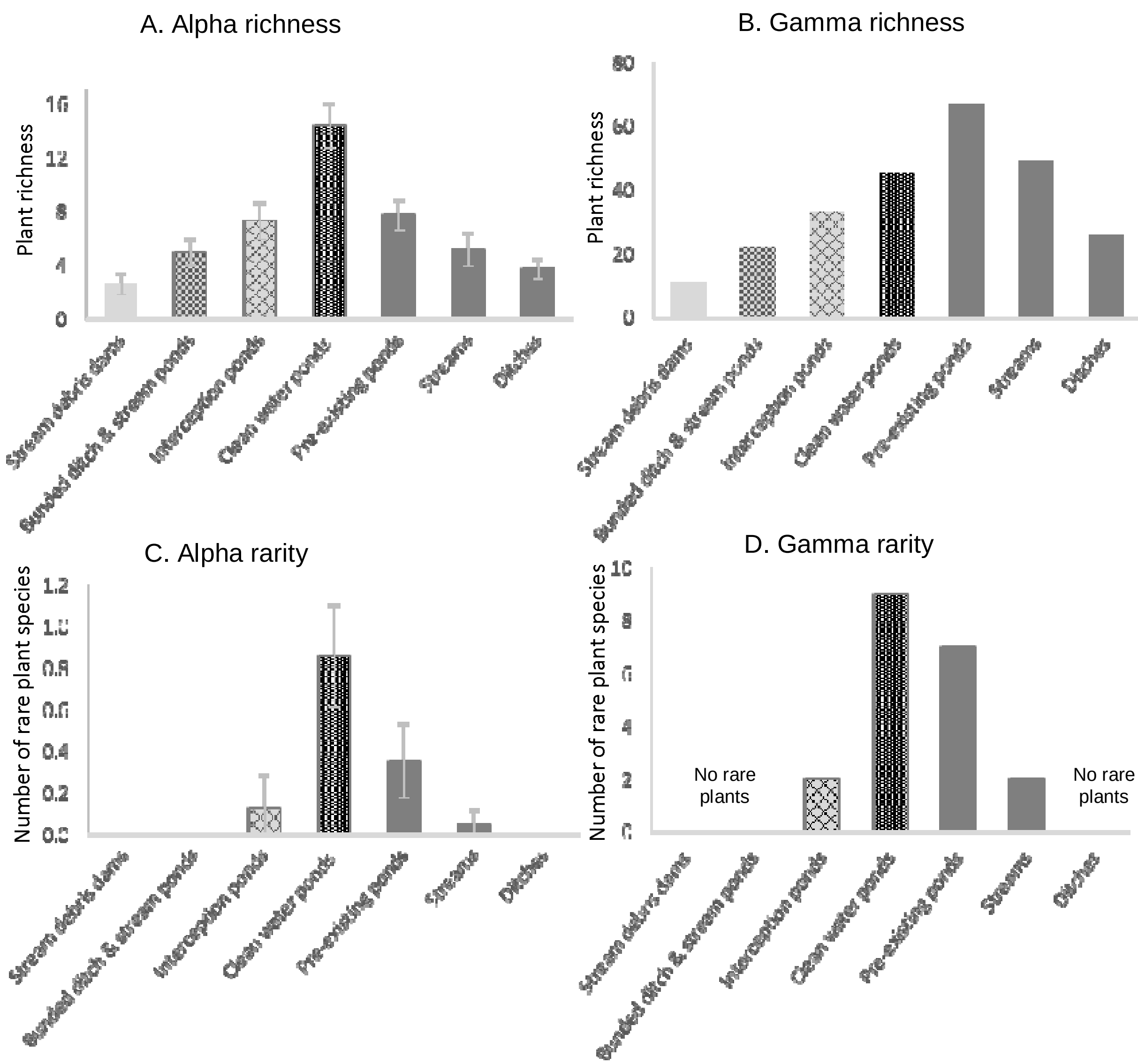

Figure 4. Alpha and gamma richness and rarity of measures ponds and other waterbody types in the Eye and Stonton catchments in 2018, when the measures were five years old. Error bars for alpha richness and rarity show 95% CLs.

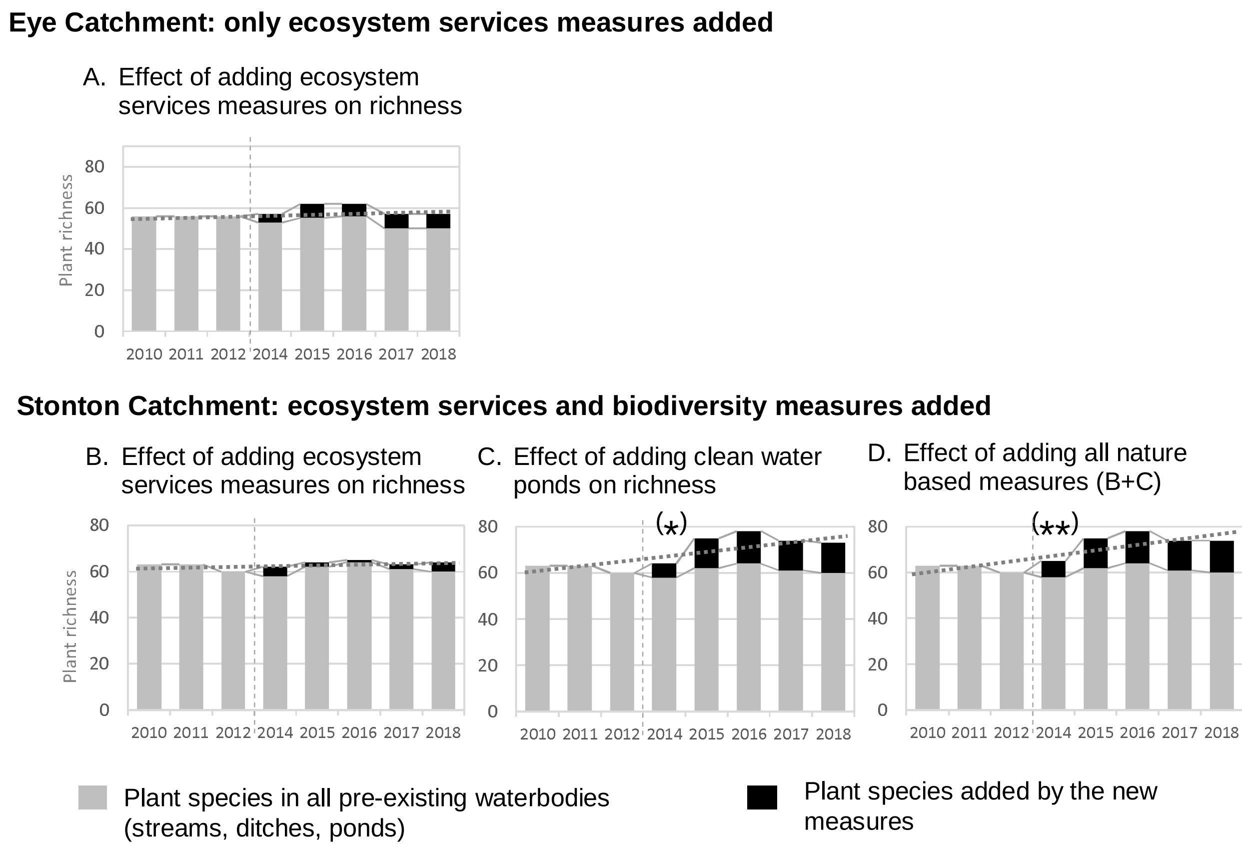

Figure 5. Change in gamma richness as a result of adding nature based measures to the experimental catchments. Ecosystem-services measures were added to the Eye Catchment. The Stonton Catchment received both ecosystem-services measures and clean water ponds. Dashed vertical lines separate the pre- and post-measure phases. Dotted lines show the simple linear regression for total plant richness. Asterisks denote the statistical significance of the regression equation: p-value <0.05 (**), < 0.01 (**), < 0.001 (***).

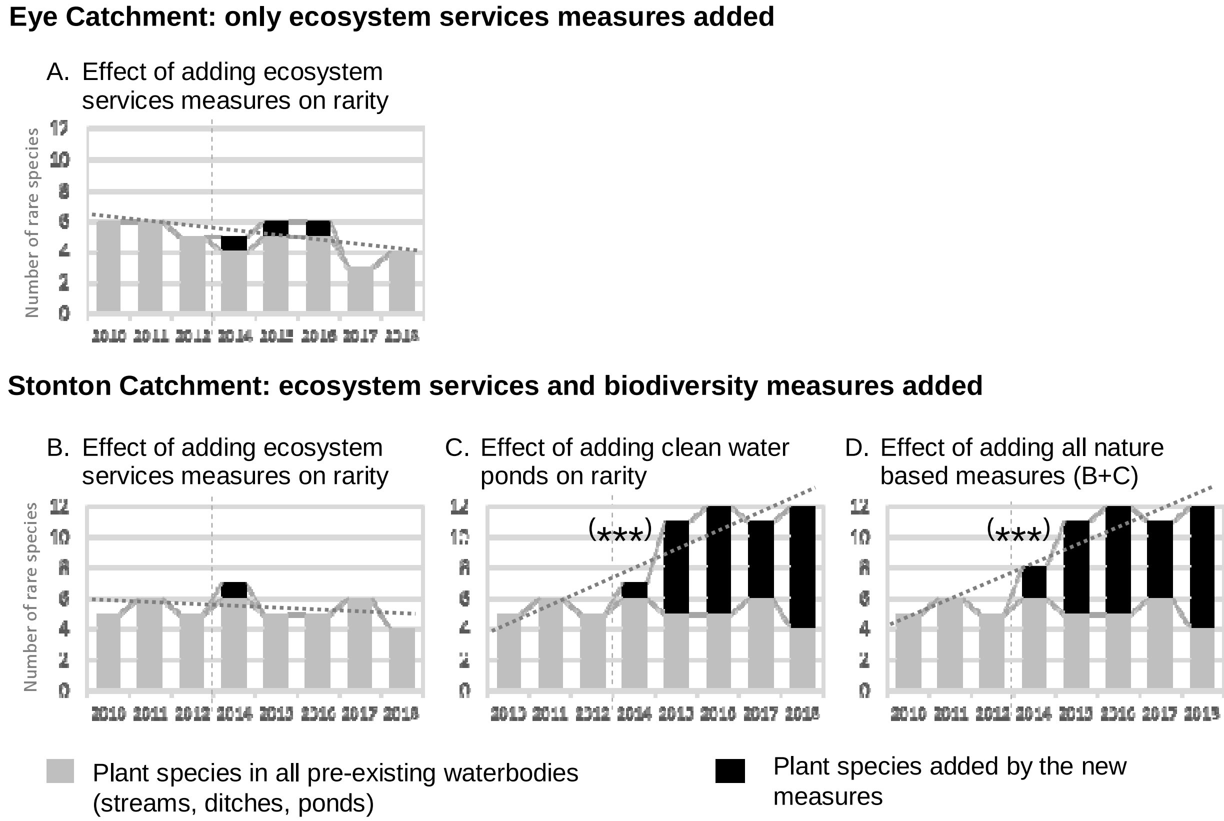

Figure 6. Change in gamma rarity as a result of adding nature based measures to the experimental catchments. Ecosystem-services measures were added to the Eye Catchment. The Stonton Catchment received both ecosystem-services measures and clean water ponds. Dashed vertical lines separate the pre- and post-measure phases. Dotted lines show the simple linear regression for total plant rarity. Asterisks denote the statistical significance of the regression equation: p-value <0.05 (**), < 0.01 (**), < 0.001 (***).