Science of the Total Environment 581–582 (2017) 105–116

Contents lists available at ScienceDirect

Science of the Total Environment

journal homepage: www.elsevier.com/locate/scitotenv

Finding clean water habitats in urban landscapes: professional researcher vs citizen science approaches

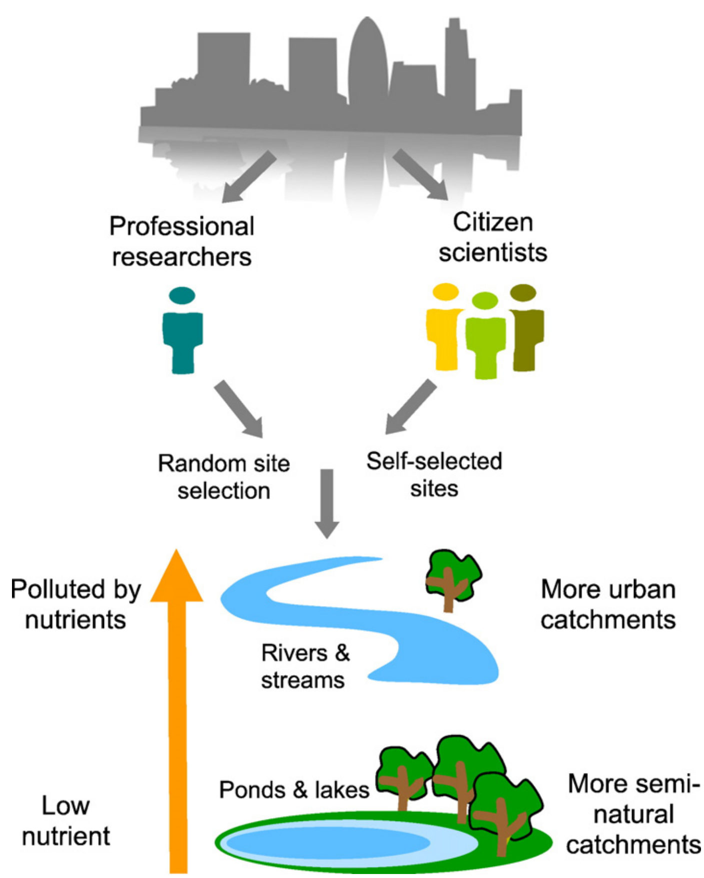
Elaine McGoff, Francesca Dunn, Luis Moliner Cachazo [i], Penny Williams *, Jeremy Biggs, Pascale Nicolet, Naomi C. Ewald

Freshwater Habitats Trust, Bury Knowle House, North Place, Headington, Oxford OX3 9HY, UK

HIGHLIGHTS

- Citizens and ecologists collected nutrient data from waterbodies across London.
- Citizen sites were self-selected, the ecologists collected stratified random data.
- Both showed that ponds & lakes had fewest nutrients, rivers were far more polluted.
- However citizen nutrient data failed to show expected relationships with land cover.
- Citizen science data would be more valuable if survey locations were pre-selected.

GRAPHICAL ABSTRACT



* Corresponding author.
E-mail addresses: info@freshwaterhabitats.org.uk, pwilliams@freshwaterhabitats.org.uk (P. Williams).
[i] Present address: Imperial College London, SW7 2AZ, UK.

http://dx.doi.org/10.1016/j.scitotenv.2016.11.215

ARTICLE INFO

Article history:
Received 25 August 2016
Received in revised form 12 November 2016
Accepted 30 November 2016
Available online 7 January 2016

Keywords:
River
Stream
Pond
Ditch
Lake
Nutrients

ABSTRACT

This study investigated patterns of nutrient pollution in waterbody types across Greater London. Nitrate and phosphate data were collected by both citizen scientists and professional ecologists and their results were compared. The professional survey comprised 495 randomly selected pond, lake, river, stream and ditch sites. Citizen science survey sites were self-selected and comprised 76 ponds, lakes, rivers and streams. At each site, nutrient concentrations were assessed using field chemistry kits to measure nitrate-N and phosphate-P.

The professional and the citizen science datasets both showed that standing waterbodies had significantly lower average nutrient concentrations than running waters. In the professional datasets 46% of ponds and lakes had nutrient levels below the threshold at which biological impairment is likely, whereas only 3% of running waters were unimpaired by nutrients. The citizen science dataset showed the same broad pattern, but there was a trend towards selection of higher quality waterbodies with 77% standing waters and 14% of rivers and streams unimpaired.

Waterbody nutrient levels in the professional dataset were broadly correlated with landuse intensity. Rivers and streams had a significantly higher proportion of urban and suburban land cover than other waterbody types. Ponds had higher percentage of semi-natural vegetation within their much smaller catchments. Relationships with land cover and water quality were less apparent in the citizen-collected dataset probably because the areas visited by citizens were less representative of the landscape as whole.

The results suggest that standing waterbodies, especially ponds, may represent an important clean water resource within urban areas. Small waterbodies, including ponds, small lakes < 50 ha and ditches, are rarely part of the statutory water quality monitoring programmes and are frequently overlooked. Citizen scientist data have the potential to partly fill this gap if they are co-ordinated to reduce bias in the type and location of the waterbodies selected.

1. Introduction

The process of urbanisation is an ongoing global phenomenon affecting both the developed and developing worlds. Current projections estimate that the extent of urban land cover worldwide will increase by 185% between 2000 and 2030 (Seto et al., 2012), with concomitant infill-development increasing building density and decreasing the remaining extent of urban green space (Gledhill et al., 2008).

Urbanisation has been shown to cause profound changes to the freshwater environment: rivers and streams are typically channelised or culverted whilst most standing waters are either destroyed or modified into amenity features (Booth and Jackson, 1997; Meyer and Wallace, 2001; Wood et al., 2003). Hydrological changes alter the availability of water including its volume, velocity and periodicity, which in turn impacts water chemistry, sediment loading and the character of bottom substrates (Boyer and Polasky, 2004). The run-off to waterbodies from urban surfaces can be polluted by a combination of elements including oils, metals, nutrients, pathogens and a wide range of man-made compounds: an issue which is compounded, particularly in running waters, by inputs of treated and untreated sewage, licensed and unlicensed industrial discharges and effluents that reach watercourses as a result of drainage system misconnections (Gerken Golay et al., 2013; Latimer and Quinn, 1998; Lenat and Crawford, 1994; Paul and Meyer, 2001; Sonoda et al., 2001). This plethora of physico-chemical changes inevitably impacts freshwater biodiversity and biological processes, with most studies suggesting that the net effect is strongly detrimental (Booth et al., 2004; Lenat and Crawford, 1994; Paul and Meyer, 2001, and references therein).

Given that the impacts of urbanisation on freshwaters are held to be wide-ranging and generally damaging, it is surprising that there are remarkably few empirical data describing the quality of freshwaters in urban areas. In rural landscapes, studies have shown considerable heterogeneity in the extent to which waterbodies degrade as a result of anthropogenic impacts. Small waterbodies like ponds, for example, have sometimes been shown to retain relatively clean water and high biodiversity even in intensively managed agricultural catchments, enabling them to contribute disproportionately to regional biodiversity (Williams et al., 2004; Davies et al., 2008b; Biggs et al., 2016a, 2016b). There are no equivalent studies that compare waterbody types in urban environments, despite the multiple ecosystem services urban freshwaters provide including flood amelioration, water treatment, delivery of potable water, protection of biodiversity, creation of amenity resources and provision of green space with its inherent value for promoting emotional and physical health (Hassall, 2014 and references therein; Hassall and Anderson, 2015; Völker and Kistemann, 2015; Bradley and Frost, in this issue). Increasing our understanding of the value of the urban freshwater resource has the potential to enable us to better balance and protect these uses.

In Europe, the ecological quality of freshwaters is monitored under the auspices of the EU Water Framework Directive (2000/60/EC) which requires member states to maintain the quality of all fresh waters across their territory. In practice, only a tiny proportion of the freshwater network is assessed in any EU State and, statutory monitoring for the Directive has a strong bias towards larger waters: focusing on rivers and lakes over 50 ha. This means that small streams, headwaters, ditches, ponds and most lakes are almost entirely overlooked both in terms of monitoring, and action to protect their quality.

A possible solution to the paucity of information about the quality of urban freshwaters would be to augment professional water quality monitoring data with citizen science-collected data. Citizen-collected data are already essential for many disciplines involving the collection of large-scale field datasets, and are beginning to be used for freshwaters particularly for assessing the river quality in order to pick-up pollution events (Canfield et al., 2002; Loperfido et al., 2010; Obrecht et al., 1998; Rotman et al., 2012). Such an approach has the added benefit of directly involving communities in activities to protect their local environment, and is particularly feasible for urban areas because of the large audience of potential volunteers (Canfield et al., 2002).

In the current study our aims were twofold:

(i) to evaluate for the first time the patterns of water quality, evident across all fresh waterbody types within catchments of differing levels of urbanisation in a major city, Greater London, using data collected by professional cologists
(ii) to establish whether citizen science-collected data has the potential to adequately replicate the patterns evident in a professionally collected research dataset.

To assess water quality we focused on two widespread pollutants: phosphate and nitrate. These nutrients are amongst the most pervasive sources of freshwater pollution globally. They directly result in the over growth of aquatic macrophytes and algae and the suppression of less tolerant taxa causing, in turn, a raft biological, health and economic impacts, including loss of plant, invertebrate and fish diversity, declines in the aesthetic and amenity value of freshwater, and in some cases the development of toxic blue-green algae blooms that are harmful if ingested by humans and animals. As a result of their wide ranging effects phosphate and nitrate are amongst the most widely used metrics for assessing ecological quality in international monitoring programmes such as the Water Framework Directive (Liu et al., 2012; Brahney et al., 2015; Mekonnen and Hoekstra, 2015).

2. Material and methods

2.1. Site selection and classification

For the professionally collected research dataset our aim was to get a representative indication of nutrient levels in the different waterbody types across the London area. The survey area was defined as the zone encompassed by the M25 motorway which encircles Greater London, and includes both London's core commercial districts and its suburban periphery; an area with a total population of just over 8.5 million people. Initial map analysis and ground-truthing suggested that five main waterbody types occurred within this area: ponds, lakes, rivers, streams and ditches. To ensure good coverage, a stratified random sampling approach was used, with the aim to being to survey 100 locations for each waterbody type, giving an overall density of 12 waterbodies per 1 km square.

Waterbody types were defined using the criteria listed in Table 1. For each waterbody type, individual sampling locations were identified in a two stage process (i) 1 km squares were selected at random with replacement, using Ordnance Survey GIS layers (ii) within each grid square, sampling sites were selected at random from areas that were easily accessible. The survey did not include waterbodies in areas of curtilage e.g. garden ponds. If the waterbody type being selected was not present in the square or in an accessible area, another square was randomly selected. Survey sites were sampled by two professional ecologists in autumn from September to early December. Half were surveyed in 2014, half in 2015.

Citizen scientists were recruited by the charity Earthwatch for the FreshWater Watch project. Their objective was to test water quality in local waterbodies to contribute to a database aiming to assess the health of freshwater ecosystems on a global scale. To participate in FreshWater Watch each volunteer undertook a day's training as Citizen Science Leaders during which they were trained in water chemistry testing methods, and asked to undertake water chemistry tests at local waterbodies in the following months. The sampling started in April of 2013, and is ongoing, although for our purposes we used a cut-off at the end of March 2016 for compiling the dataset. Citizen Science Leaders have sampled a range of areas in the UK, but we used a subset of those falling within the Greater London area for comparison with our professional research dataset. A total of 36 Citizen Science Leaders collected nutrient data within this area.

Table 1
Definitions of aquatic habitats used in the survey.
Modified from Williams et al. (2004).

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

2.2. Nutrient sampling

In the current study, both the professional research and citizen science surveys, measured nutrients with easy-to-use nitrate and phosphate field kits with a relatively low minimum detection threshold (0.2 mg/l and 0.02 mg/l respectively). We used Kyoritsu PackTest water chemistry kits which provide an in-field colourimetric reaction for unfiltered water samples, which is then compared with a colour chart to indicate nutrient level categories. The reaction takes place when the water sample is combined with pre-measured reagents in a closed sample tube. The phosphate kits measure phosphate-P with a detection level between 0.02 and 1 mg/l, and are based on an inosine enzymatic method (Berti et al., 1988), using 4-Aminoantipyrine. The colour change in the sample tube is compared visually with a six point colour chart. Values were recorded as the range value between the two closest colour matches (Table 2). The nitrate-N kits have a detection range of 0.2–10 mg/l and the reaction is based on the *N*-(1-napthyl)-ethylenediamine method, using a zinc reduction for converting nitrate to nitrite. As with the phosphate-P kits, the colour reaction is compared with a six point colour chart (Table 2).

2.3. Analysis

In order to provide an overview of London's water quality, samples were classified into three categories depending on their nitrate-N and phosphate-P levels: clean, some nutrient pollution and nutrient polluted. The clean category was defined as those sites falling below the thresholds of 0.05 mg/l phosphate-P and 1 mg/l nitrate-N. The 'some nutrient pollution' category included sites which fell into the 0.05–0.1 mg/l phosphate-P category and for nitrate the 1–2 mg/l nitrate-N category. The 'nutrient polluted' category was all values above this.

Thresholds were based on the concept that clean water should have a chemistry and biology which is normal for a given area in the absence of human disturbance (Williams et al., 2010). This is commonly referred to as 'the reference condition', 'minimally impaired water quality' or 'natural background levels' and, in Europe it is equivalent to 'High' status in the Water Framework Directive (WFD). For phosphorus, clean water thresholds for the rivers, streams and lakes were related to the values used to define High status in WFD (Carvalho et al., 2006; UKTAG, 2013). For ponds and ditches, where no regulatory standards have been defined for phosphorus, empirical data sources were used to identify reference conditions of nutrients (Biggs et al., 2005). For nitrate, there are no ecologically relevant regulatory standards defined for any freshwater habitats in the UK (WFD standards are related to drinking water and are much higher than the values known to have ecological impacts). Consequently a range of literature sources were used to assess the nitrate levels associated with sites which are not exposed to anthropogenic nutrient inputs or which support unimpaired biotic

Table 2
Colour categories as defined by the Kyoritsu PackTest kits.

Category	Nitrate-nitrogen mg/l range (midpoint)	Phosphate-phosphorus mg/l range (midpoint)
1	0–0.2 (0.1)	0–0.02 (0.01)
2	0.2–0.5 (0.35)	0.02–0.05 (0.035)
3	0.5–1 (0.75)	0.05–0.1 (0.075)
4	1–2 (1.5)	0.1–0.2 (0.15)
5	2–5 (3.5)	0.2–0.5 (0.35)
6	5–10 (7.5)	0.5–1 (0.75)

assemblages (Dodds et al., 1998; Biggs et al., 2005; González Sagrario et al., 2005; James et al., 2005; Lambert and Davy, 2011; Moss et al., 2013).

To provide a combined classification, using both nitrate-N and phosphate-P, we adapted the one-out-all-out rule used by the Water Framework Directive. A clean waterbody was defined as falling below the clean water threshold for both nitrate-N and phosphate-P. Waterbodies with only one determinand falling below the threshold were categorised as having 'some nutrient pollution'. Waterbodies were categorised as 'nutrient polluted' when both nitrate-N and phosphate-P levels were higher than the clean water threshold.

The midpoint value for each nitrate-N and phosphate-P range (Table 2) was used for statistical analysis. Because the values were categorical, non-parametric statistics were used throughout (Rank ANOVA and Spearman rank correlations). Statistical analyses were carried out using the R package version 3.2.3 (R Core Development Team) using standard base packages.

2.4. Deriving catchment land cover

The surface water catchment of streams and rivers was modelled using GIS Drainage Network Analysis in ArcMap version 10.3.3. A digital elevation model (DEM) was generated using data supplied under open license by the Environment Agency (https://data.gov.uk/dataset/lidar-tiles-tile-index). Digital Terrain Modelling data was downloaded at 2 m spatial resolution for the whole of the London area, and was then resampled and rasterised into a 25 m DEM raster. The Fill tool in ArcMap was used to correct for any potential smaller errors in the DEM, followed by use of the Flow Direction tool, which created a flow direction grid on the filled DEM. The Flow Accumulation tool was used to give the drainage area for every pixel in the filled DEM. These layers were then used to delineate the watersheds for the larger waterbodies.

The drainage network analysis was successful for rivers, streams and some lakes, although in some areas of London no Lidar data were available. In this case, and in cases where the watershed was abruptly truncated, a combination of the ArcMap basemaps and the modelled flow network was used to delineate the catchment area.

For ponds and ditches which rarely register within the DEM model, the waterbodies were buffered, using circle with a radius of 100 m from the sampling point outwards as a proxy for catchment area.

Land Cover Map 2007 data (Morton et al., 2011) were used to determine landuse in the catchments. The 25 m resolution landcover data are broken into 23 land classes, and the catchments we delineated were overlaid on this landcover dataset, to clip out the landuse. The area of each of the land cover types within the clipped area was then calculated. For analysis, the 23 classes were simplified into five broad groups representing a scale of urbanisation, from semi-natural to urbanised catchment. These were: i) semi-natural which included broadleaved and coniferous woodland; rough, neutral and acid grassland; heather; heather grassland; inland rock; freshwater; salt water; littoral sediment and rock; and salt marsh, ii) arable, iii) improved grassland, iv) suburban and v) urban.

3. Results

3.1. Data collected and its geographical spread

Of the planned 500 sites, 495 were ultimately sampled for the professional research dataset, 99 sites for each waterbody type. The spatial distribution of sites (Fig. 1) shows a paucity of sample sites in the south of the survey area which is dominated by the dry chalk landscapes of the North Downs. Ditch sites were predominantly located around the periphery of the survey area, because ditches were largely absent from the heavily built up areas of central London.

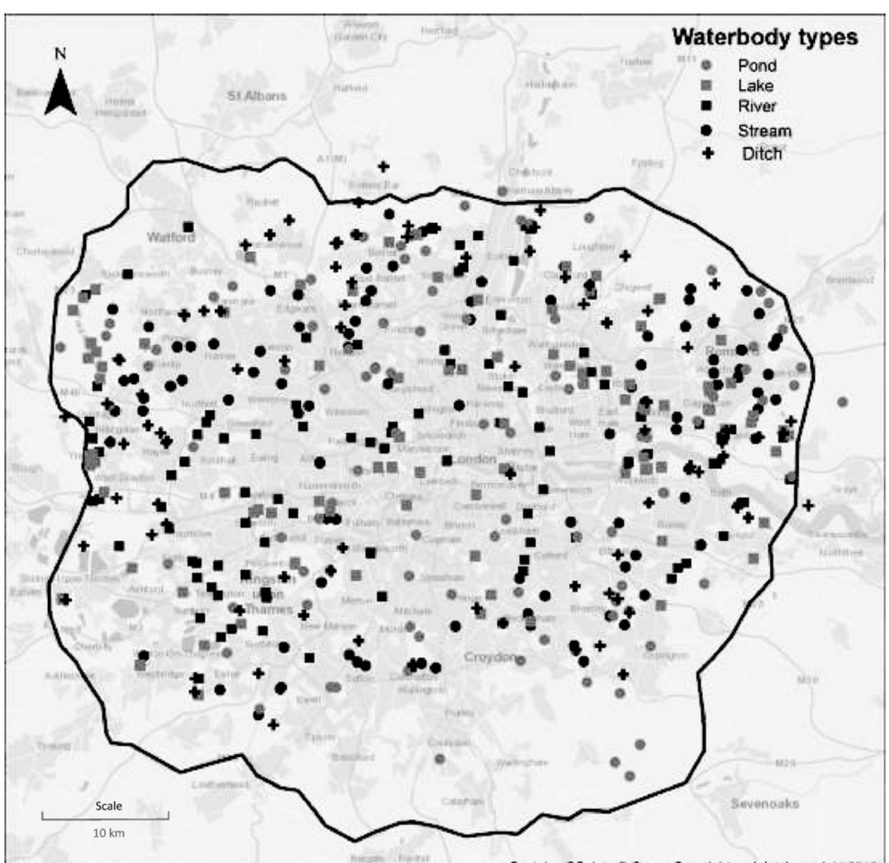


Fig. 1. Professional sample sites for each waterbody type within the Greater London area, with the M25 orbital motorway highlighted in black. N = 99 of each waterbody type.

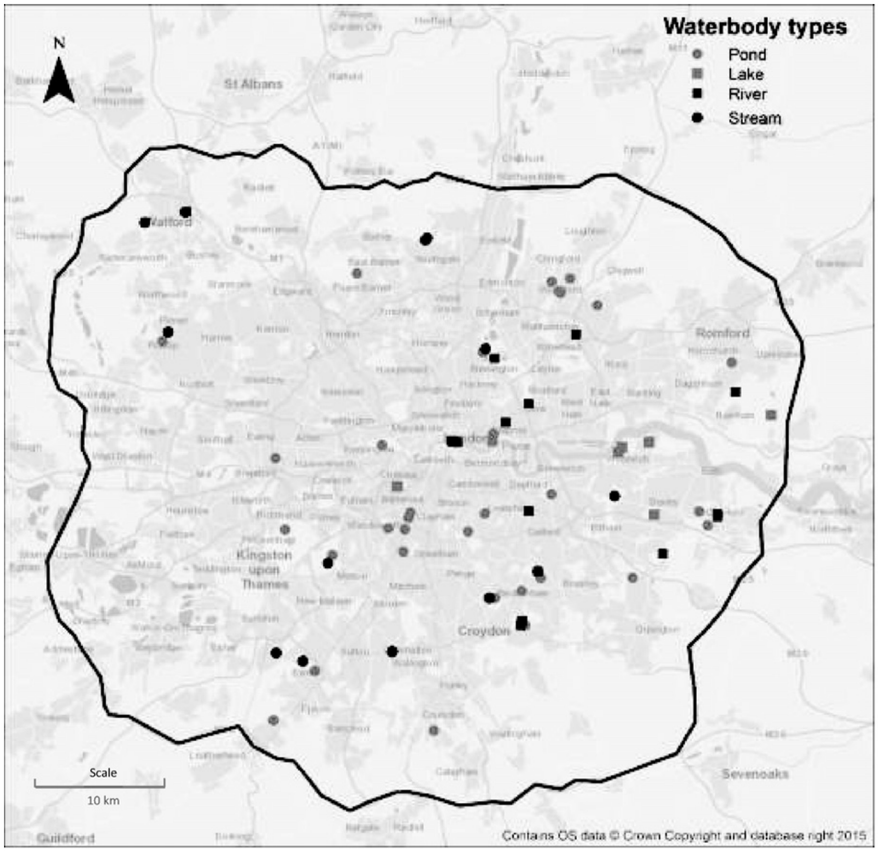


Fig. 2. Citizen science sample sites for each waterbody type within the Greater London area, with the M25 orbital motorway highlighted in black. N = 76 (ponds = 40, lakes = 7, rivers = 16 and streams = 13).

Citizen scientists self-selected a total of 76 waterbodies: 40 ponds, 7 lakes, 16 rivers and 13 streams. Ditches were not chosen for sampling by volunteers. The data points were reasonably well spread across the Greater London area (Fig. 2), but there was a paucity of sites in the north west quarter of London (12.4%) compared to the northeast and southeast quarters (34.8% and 31.5% respectively).

There were no clear spatial trends in water quality across either the professional research or citizen science datasets as a whole. Both nutrient polluted and clean water sites were relatively evenly spread across the study area, with low-nutrient waterbodies found even within the very centre of the city (Figs. 3 and 4).

3.2. Differences between waterbody types

In both the professional researcher and the citizen science datasets, standing waterbodies had lower nutrient levels than running waters (Table 3).

Within the professional research dataset, there were significant differences in levels of both nitrate-N and phosphate-P amongst the waterbody types (rank ANOVA, $p \leq 0.001$). Overall ponds and lakes were the cleanest waterbodies in terms of nutrient levels. For nitrate-N the trend in median values was ponds ≈ lakes < ditches < streams < rivers (Fig. 5). Posthoc tests showed that the differences between all waterbody pairs except ponds and lakes were significant (Tukey HSD, $p < 0.01$). For phosphate-P the general trend in mean values was ponds < lakes < ditches ≈ rivers < streams, but the only significant difference was that rivers and streams had significantly higher phosphate levels than other waterbodies (Tukey HSD, $p \leq 0.01$).

Amongst the five waterbody types in the professional dataset, only ponds had median nutrient levels that were low enough to fall below the clean-water threshold for both nitrate and phosphate (0.1 mg/l, 0.04 mg/l respectively). The median for lakes and ditches was below this threshold for nitrate (0.35 mg/l, 0.75 mg/l respectively), but not phosphate (0.08 mg/l, 0.15 mg/l). Streams and rivers showed evidence of extensive nutrient pollution, with medians typically well above the clean-water threshold for both nutrients (nitrate: 3.5 mg/l, 7.5 mg/l; phosphate: 0.35 mg/l, 0.15 mg/l respectively) (Fig. 5).

Data gathered by the citizen scientists also showed a significant difference between waterbody types for both nitrate-N and phosphate-P (rank ANOVA, $p \leq 0.01$). For nitrate-N the general trend in median values was ponds ≈ lakes < streams < rivers, with rivers and streams having significantly higher nitrate-N than lakes and ponds (Tukey HSD, $p \leq 0.01$). The same general pattern was evident for phosphate-P (ponds ≈ lakes < streams ≈ rivers) with a significant difference between the standing and running waters (Tukey HSD, $p \leq 0.05$) (Fig. 6). However, mean levels of phosphate-P were much lower in the citizen science dataset than the professional research dataset, with the majority of rivers and streams from the citizen science dataset falling below the clean water threshold (Figs. 5 & 6).

3.3. Relationships between nutrient levels and land cover

Across the professional research dataset as a whole there was, as expected, a relationship between waterbody nutrient levels and the land cover in the waterbodies catchment (Table 4). The proportion of semi-natural land cover was negatively correlated with both the nitrate-N and phosphate-P levels (Spearman rank correlation, $r = -0.213$, $p \leq 0.001$ for phosphate-P; $r = -0.192$, $p \leq 0.001$ for nitrate-N). Conversely, there was a positive relationship between nitrate-N and phosphate-P levels and the percentage of both urban and suburban land cover in the catchment (urban land cover: Spearman rank correlation, $r = 0.259$, $p \leq 0.001$ for phosphate; $r = 0.315$, $p \leq 0.001$ for nitrate;

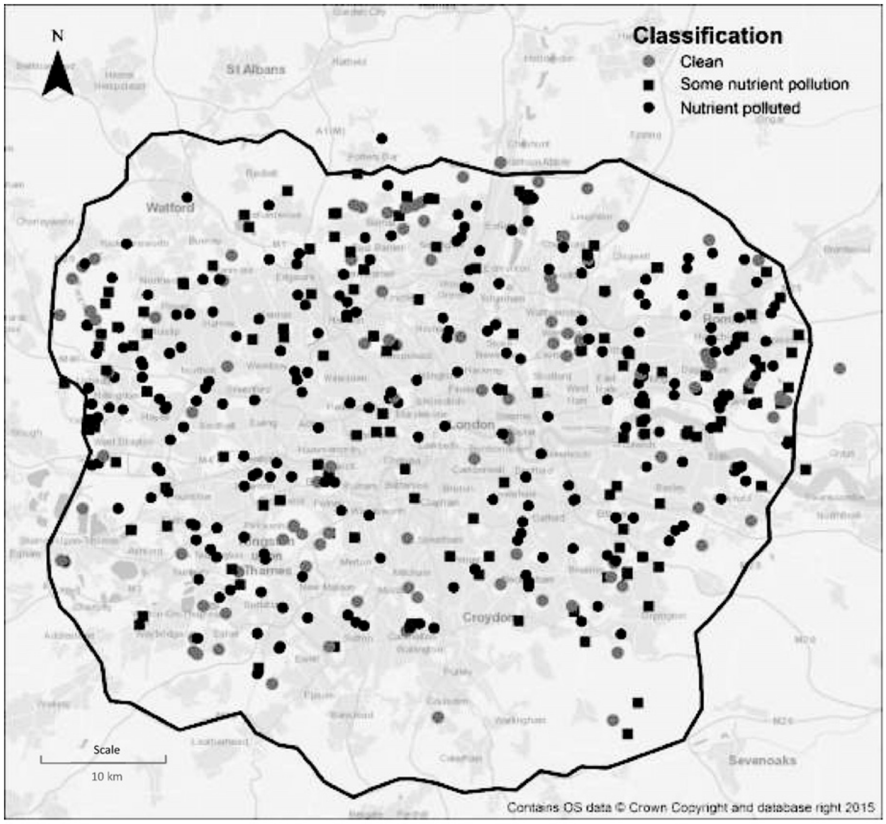


Fig. 3. Spread of sites with clean water, some nutrient pollution and nutrient pollution within the professional dataset across the study area (N = 495), with the M25 orbital motorway highlighted in black.

suburban land cover r = 0.259, $p < 0.001$ for phosphate-P, r = 0.286, $p < 0.001$ for nitrate-N).

In the professional dataset, the five waterbody types differed significantly in the proportions of urban, suburban, improved grassland and semi-natural land cover in their catchments (rank ANOVA $p < 0.01$ for al., Table 4). Posthoc analysis showed that rivers had significantly more urban land cover in their catchment than all other waterbodies; rivers and streams both had significantly more suburban land cover than ponds, ditches and lakes; ditches had significantly improved grassland more than other waterbody types. Ponds were significantly more likely to have semi-natural catchments than rivers, streams and ditches; and lakes were significantly more likely to have semi-natural land cover in their catchments than rivers (Tukey HSD, $p < 0.001$ for all) (Fig. 7).

The citizen science datasets showed fewer relationships between nutrient levels and catchment land cover (Table 4). Across the dataset as a whole, only the proportion of arable land cover showed a significant positive correlation with nitrate-N or phosphate-P (Spearman rank, $p < 0.05$, r = 0.531 and $p < 0.05$, r = 0.281 respectively). Post hoc tests to identify differences between waterbody types indicated that rivers had significantly more arable land in their catchments than ponds (Tukey HSD, $p < 0.05$) (River median = 4.5%, pond median = 0%). However, the overall percentage of arable land cover in both the professional dataset and the citizen science dataset was negligible (median = 0% across waterbodies for both datasets).

A side-by-side comparison of catchment land cover in the professional and the citizen science datasets showed significant differences (Fig. 8). Citizen science sites were significantly more suburban (Tukey HSD, $p < 0.001$; median 47.6% for citizen science and 27.9% for professional waterbodies). The professional sites had a higher proportion of improved grassland in their surrounding land cover (Tukey HSD, $p < 0.001$; median 11.7% for citizen science and 20.7% for professional waterbodies).

4. Conclusion and discussion

4.1. Nutrient pollution, land cover and waterbody type

In the current study the professional research dataset provides a snap-shot of nutrient levels across all major waterbody types within the large urban and suburban sprawl of Greater London. Remarkably few previous studies have systematically examined the freshwater resource across urban areas with most being restricted to a single waterbody type: mainly streams or rivers (Moore and Palmer, 2005; Paul and Meyer, 2001; Smith and Lamp, 2008), and more recently ponds or lakes with a focus on water retention systems (Briers, 2014; Koperski, 2010; Steele and Heffernan, 2014, Bradley and Frost, in this issue). Our findings show the importance of a broader approach revealing, in London at least, a clear dichotomy between the high nutrient-pollutant loading in running waters across the city, and the much lower nutrient levels typical of standing waters. This was particularly evident for ponds which largely retained nutrient levels below those at which significant biological degradation would be expected. Perhaps surprisingly, ponds and lakes helped to retain pockets of low-nutrient water even in the very centre of London.

Anthropogenic landuse impacts are well-established as the principle driver of freshwater degradation (Booth et al., 2004; Boyer and Polasky, 2004; McKinney, 2008; Roach et al., 2008), and urbanisation is known to have particularly degrading effects on water quality, including nutrient pollution (Donohue et al., 2005; Meybeck, 1998; Moore and Palmer, 2005; Sonoda et al., 2001; USGS, 1999, Zhang et al., in this issue). Our findings concur with this generality: showing that, across the professional dataset as a whole, waterbodies with higher nutrient levels had a significantly higher proportion of urban and suburban land cover in their catchment.

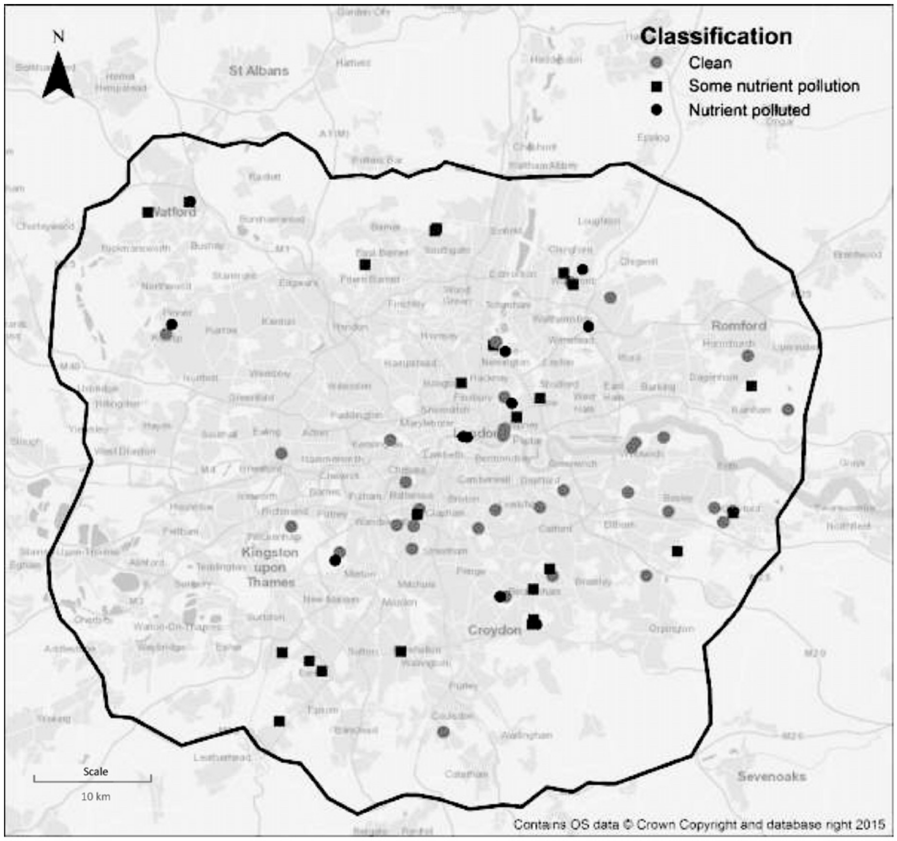


Fig. 4. Spread of sites with clean water, some nutrient pollution and nutrient pollution within the citizen science dataset across our study area (N = 76), with the M25 orbital motorway highlighted in black.

When broken down by waterbody type, the data show that the poorer water quality seen in rivers and streams was associated with a higher mean percentage of urban and suburban land cover in their catchments. Ponds and lakes, which were less nutrient polluted, typically had catchments with a larger proportion of semi-natural land cover.

It is important to note that, in the current study where the two factors waterbody type and land cover are correlated, it is not possible to definitively state the extent to which either explains the observed nutrient levels. However, the highly significant relationship between land cover intensity and both nitrate and phosphate levels across all waterbodies combined indicates the importance of landuse. The findings also concur with the literature, where highly predictive relationships between catchment landuse and waterbody nutrient levels are well established for both standing and running waterbody types (Beaulac and Reckhow, 1982; Heathwaite et al., 2003; Johnes et al., 1996).

Waterbody catchment size is likely to be an important factor explaining the between-waterbody nutrient differences in Greater London. In the current study we could not definitively measure the catchment area for all waterbodies because there were insufficient topographic data for most ditches and ponds. However, consistent relationships between catchment size and waterbody type have been established elsewhere, including in our own study of a rural landscape in the Coleshill area to the west of London. Here, rivers (average catchment area 43,850 ha), had catchment areas that were typically three orders of magnitude larger than ponds. Lake, ditch and stream catchments fell between these two extremes in order of increasing catchment size (Davies et al., 2008a). In managed landscapes, catchment area has the potential to strongly influence the extent of intensive land cover within a waterbody's catchment. Thus in intensively managed lowland England, the considerable land areas drained by rivers and streams means that there is usually no escape from pollution: somewhere in their catchments these watercourses will inevitably drain nutrient-exporting land uses. Our data for London rivers shows that their catchments include a particularly high proportion of urban and suburban land cover. The impact of even relatively small areas of urbanised land cover (0.03%–15%) has been shown to significantly degrade river and stream water quality (Allan, 2004; Donohue et al., 2006; Morse et al.,

Table 3
Percentage of waterbodies in the professional and citizen science datasets which were clean (i.e. were within the clean water threshold), had some nutrient pollution and were nutrient polluted.

Waterbody type	Number of waterbodies (n)		% clean		% some nutrient pollution		% nutrient polluted	
	Professional	Citizen science	Professional	Citizen science	Professional	Citizen science	Professional	Citizen science
Ponds	99	40	54.5	72.5	41.4	22.5	4.0	5.0
Lakes	99	7	36.4	100	41.4	0	22.2	0
All standing waters	**198**	**47**	**45.5**	**76.6**	**41.4**	**19.1**	**13.1**	**4.3**
Rivers	99	16	0	12.5	6.1	56.3	93.9	31.3
Streams	99	13	6.1	15.4	15.2	61.5	78.8	23.1
All running waters	**198**	**29**	**3.1**	**13.8**	**10.7**	**58.6**	**86.4**	**27.6**
Ditches	99	0	17.2	0	37.4	0	45.5	0

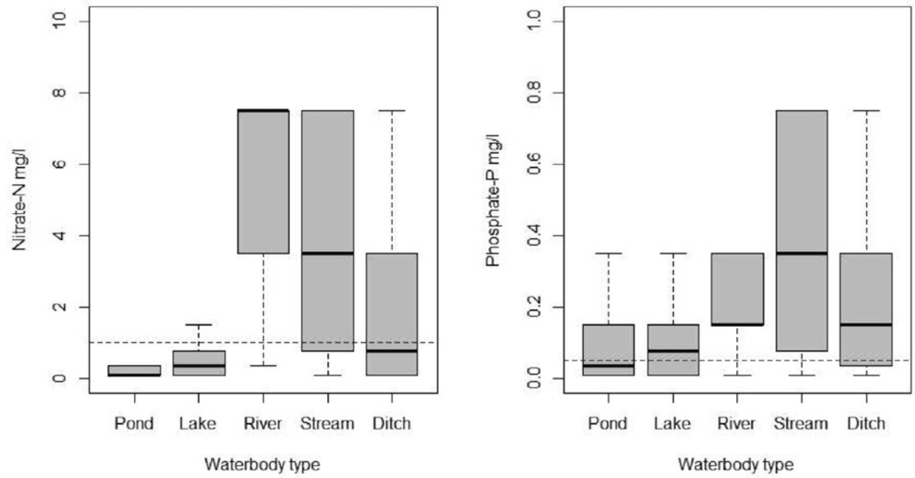


Fig. 5. Comparison of nitrate-N and phosphate-P (mg/l) levels across waterbodies for the professional dataset (n = 99 for each waterbody type) with clean water threshold indicated by dashed grey line. The dark mid line represents the median, the box represents the interquartile range, and the whiskers the minimum and maximum values, excluding outliers.

2003; Ourso and Frenzel, 2003; Roy et al., 2003; Wang et al., 1997). Hence it is unsurprising that these running waters draining large catchment areas are heavily nutrient polluted in London.

Ponds and small lakes in contrast, typically have much smaller catchment areas. This confers a greater chance that at least some will drain catchments that are entirely non-intensive, improving their potential to retain clean water. In the current study, ponds and to a lesser extent lakes, were significantly more likely to have semi-natural catchments than other waterbody types: a factor likely to explain their better water quality. Ditches, followed the same pattern as rivers, streams, lakes and ponds; having intermediate catchment sizes, and intermediate land cover intensity and nutrient levels.

Because ponds, have small catchments which can easily be dominated by a single land use type – from entirely semi-natural to entirely intensive – it would be expected that pond nutrient levels would show a broader spread of values than other waterbody types at landscape level: including sites with particularly high and particularly low nutrient levels. This pattern has certainly been demonstrated in predominantly rural landscapes (Williams et al., 2004). In the current study, however, the reverse trend was evident, with the spread of nutrient values for rivers and streams far greater than that shown by ponds and lakes (Figs. 5, 6). The most likely explanation for this is, again, the particularly high proportion of semi-natural landuse in the catchment of standing waters within this urban dataset; reducing levels of water pollution in the majority of ponds, and many lakes.

That so many of London's ponds and lakes had low nutrient status water and were located in semi-natural landscapes was not only an unexpected result, but contrasts with our findings in adjacent rural areas (Williams et al., 2004). It would be interesting to see if such a trend holds true for other urban centres.

A number of implications fall out of the findings from the professional survey. Firstly, given that the absence of nutrient enrichment is a good predictor of clean water and a prerequisite for a healthy ecosystem (Baron et al., 2003), the results suggest that, in large urban areas, ponds and lakes may offer a potential refuge for freshwater biodiversity (Hill et al., 2016). This is a pattern that has already been shown to be true in agriculturally impacted rural landscapes, where ponds typically support a higher proportion of gamma diversity, and more uncommon species than rivers, streams and ditches (Davies et al., 2008b; Karaus et al., 2013; Martinez-Sanz et al., 2012; Williams et al., 2004).

The results also have implications for urban planning. There is an increasing interest in sustainable urban design and management worldwide, yet this is proving difficult to achieve in practice (Dias et al., 2014; John et al., 2015). Authors such as Gagnéa et al. have called for simple design frameworks, which can help planners that have no biological knowledge better protect biodiversity in man-made

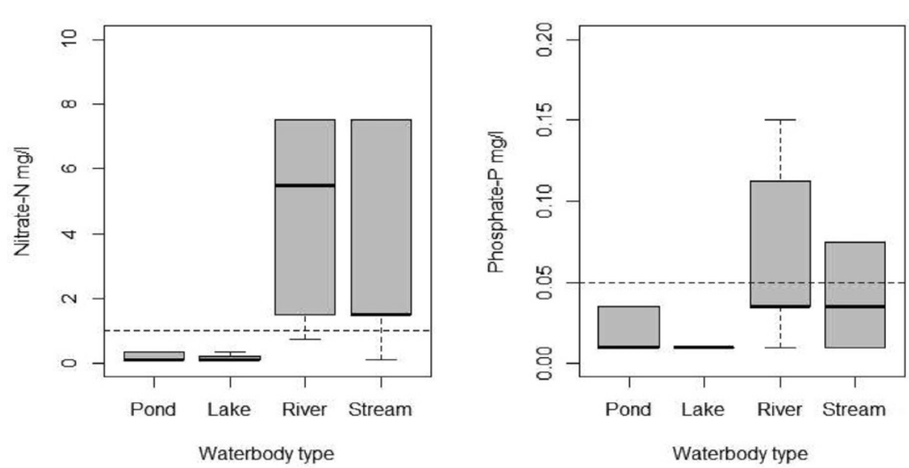


Fig. 6. Comparison of nitrate-N and phosphate-P (mg/l) levels across waterbodies for citizen science dataset (n: ponds = 40; lakes = 7; rivers = 16, streams = 13) with clean water threshold indicated by dashed grey line. The dark mid line represents the median, the box represents the inter-quartile range, and the whiskers the minimum and maximum values, excluding outliers.

Table 4
Results of Spearman rank correlations between nutrients and land cover for (a) professional ecologists (b) citizen scientists.

Landuse	Nutrient	Spearman rho	p value
(a) Professional ecologists			
Urban	Nitrate	0.315	0.001
	Phosphate	0.259	0.001
Suburban	Nitrate	0.285	0.001
	Phosphate	0.259	0.001
Arable	Nitrate	0.104	ns
	Phosphate	0.006	ns
Improved	Nitrate	−0.027	ns
	Phosphate	−0.018	ns
Semi-natural	Nitrate	−0.192	0.001
	Phosphate	−0.213	0.001
(b) Citizen scientists			
Urban	Nitrate	0.044	ns
	Phosphate	0.0584	ns
Suburban	Nitrate	0.120	ns
	Phosphate	−0.121	ns
Arable	Nitrate	0.531	0.001
	Phosphate	0.281	0.05
Improved	Nitrate	0.164	ns
	Phosphate	0.080	ns
Semi-natural	Nitrate	−0.054	ns
	Phosphate	−0.045	ns

environments (Eigenbrod, 2016; Gagnéa et al., 2015). Gagnéa et al.'s initial framework suggestions specifically highlight the importance of protecting unaltered land cover around waterbodies because of its positive effect on water quality and biodiversity. Our data suggest that it would be useful to refine this proposal. Thus, although river and stream corridors are undoubtedly important freshwater and wetland habitats, it is likely to be particularly critical to maintain high quality land cover around standing waters to protect ponds and lakes. Maintaining this focus is doubly important given scientific evidence showing how difficult it is to fully restore freshwater habitats once they have been degraded. Both lake and river restoration, for example have shown, at best, only partial recovery to their former quality over medium timescales of 20–30 years (Palmer et al., 2014; Phillips et al., 2015). For rivers in particular both empirical data and catchment modelling clearly shows that it is essentially impossible to fully restore rivers to anything like their former quality in anthropogenically impacted landscapes (Booth et al., 2004; Hering et al., 2015; Miller et al., 2010; Verdonschot et al., 2016).

4.2. Citizen science

Survey data provide the foundation that underpins our understanding of freshwater conservation and the development of policies for freshwater protection. Given the enormous number of small and large waterbodies that make up the freshwater network (in the UK, c. 500,000 ponds; c. 40,000 lakes, c. 600,000 km ditches and c. 250,000 km streams and rivers; data derived from Brown et al., 2006), it is self-evident that only a tiny proportion of the water network can ever be included in statutory surveillance programmes. However, most statutory surveillance strategies are also weakened by poorly representing the freshwater environment: focusing almost exclusively on larger waterbodies, mainly rivers, which have traditionally been seen as more important both economically and ecologically (Biggs et al., 2014). In recent years, the ecological validity of the bigger-is-better assumption has been challenged, with a range of studies beginning to show the exceptional biodiversity value of smaller waters like ditches, ponds and headwaters (Davies et al., 2008b; US EPA, 2015; Williams et al., 2004; Verdonschot et al., 2011). However this knowledge has not been translated into statutory survey or monitoring of small waterbodies in practice, not least because of the economic implications of re-organising existing survey strategies and including a greater number of waterbody types.

In the current study we evaluated whether new citizen science approaches to water testing could be applied to this problem. Specifically we wanted to find out if the less structured self-selection of sites by citizen scientists could adequately reproduce the results of a fully stratified random sampling design to representatively assess water quality in all waterbody types in urban areas.

Our results indicated that at a broad level, citizen science successfully distinguished the main trend evident in the study: showing that ponds and lakes were the least nutrient-polluted waterbodies, and were significantly cleaner than rivers and streams. We are not aware of any other studies which have attempted to use citizen science methods to generate such data with other studies generally focusing on one waterbody type – usually either streams and rivers or lakes

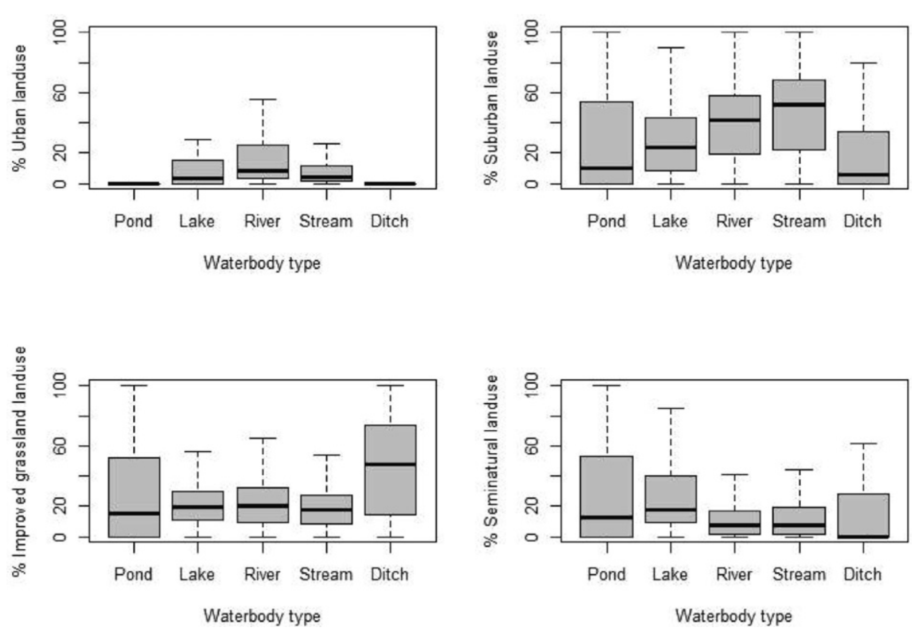


Fig. 7. Boxplots comparing landuse percentages for urban, suburban, improved grassland and semi-natural landuse across waterbody types, for professional dataset. The dark mid line represents the median, the box represents the interquartile range, and the whiskers the minimum and maximum values, excluding outliers.

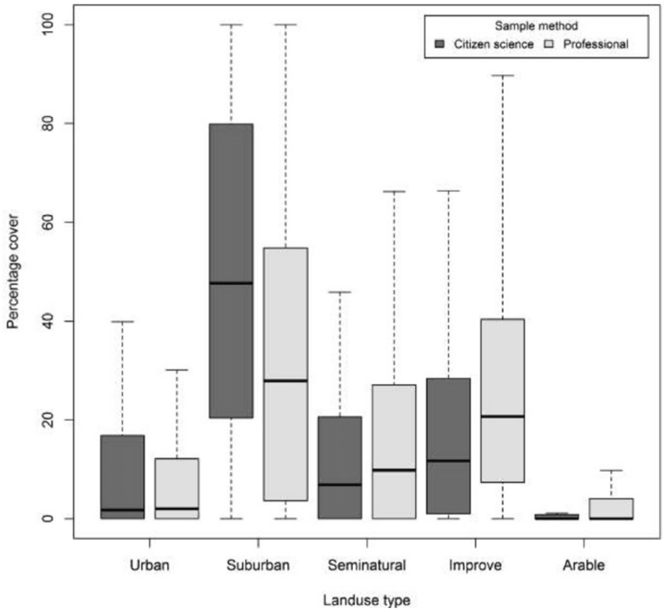


Fig. 8. Percentage landuse summed across waterbody types for the professional and citizen science watersheds. The dark mid line represents the median, the box represents the interquartile range, and the whiskers the minimum and maximum values, excluding outliers.

(Buytaert et al., 2014; Canfield et al., 2002, Lottig et al., 2014; Muneoka et al., 2014; Maas et al., 1991; Nicholson et al., 2002; Overdevest et al., 2004).

Although our citizen data were broadly successful, the results showed some disparity with the professionally collected research dataset. Phosphate-P levels recorded across the citizen science dataset were considerably lower than in the professional dataset. Waterbodies sampled by citizen scientists also tended to have higher levels of suburban land cover and, linked to this, the expected positive relationship between land cover intensity and nutrient levels, that was shown well by the professional dataset, was not detected in the citizen science dataset.

Previous comparisons of professional and citizen science collected water quality data have also identified differences in the results produced (Muenich et al., 2016; Nerbonne et al., 2008, Peckenham and Peckenham, 2014). For example, assessment of surveys of the Wabash River in Illinois, United States, concluded that citizen scientists using less sensitive nitrate and phosphate test strips than those in the present study were able to generate nitrate + nitrite-N data that generally agree with lab-determined values. However, orthophosphate results were consistently over-predicted by volunteers compared to lab values. This was partly because of limitations of the test strips which had too coarse categories to adequately reflect laboratory phosphate values (Muenich et al., 2016).

In the present study, in which professional researchers and citizen scientists used the same test kits and methods, the differences between the two sets of findings must inevitably reflect differences in data collection.

It is possible that seasonality could have played a role in generating some of the observed differences since the professional dataset was collected in autumn, whereas volunteers were free to survey at any time of year. It has long been known that both nitrate and phosphate can fluctuate seasonally, often reaching their lowest levels in late summer when these nutrients have been removed from the water and built into aquatic macrophyte and algal biomass, then climbing again in winter as biomass dies-back and decays, releasing nutrients into the water (Heron, 1961).

A second issue was that the citizen science dataset was substantially smaller than the professionally collected survey ($n = 76$ vs 495), reducing statistical confidence that the data are representative and increasing the potential for type II errors. The self-selected site approach is known to be appealing to volunteers who are usually particularly interested in discovering the quality of waterbodies in their own neighbourhood However, in this case it gave the dataset a more unbalanced design since volunteers showed a strong preference for surveying ponds rather than other waterbodies, and excluded ditches entirely. The higher levels of suburban land cover and better water quality evident in the citizen science dataset is likely to reflect the preference for volunteers to select sites that are close to home, easy to access and visually appealing; such as parks and nature reserves, rather than the more obscure locations that inevitably form part of a more stratified approach.

Overall, our findings suggest that an ad-hoc citizen science programme can provide information that broadly characterises the freshwater regime in urban areas. Given the finding that urban ponds had particularly lower nutrient levels, and that citizen scientists showed a preference for surveying these sites, it is possible that a well-directed citizen science approach could be used to supplement information used by government agencies to manage freshwaters, and may be particularly useful for baseline screening: to identify potential clean water sites which may warrant additional protection. An added benefit of citizen scientist involvement is that it also has the potential to garner greater public support for environmental protection measures for these areas (Cline and Collins, 2003; Lubell et al., 2002).

To fully replicate the results of the professional study would require a greater degree of structure and organisation than was tested in the current citizen scientist programme. There are, of course, trade-offs in adopting a more intensively managed a citizen science programme of this kind: pre-planning, training and ongoing support for volunteers has far greater staff and resource requirements, which counter balances the low-cost incentive of a citizen science approach (Thornhill et al., 2016). For the current survey, the main requirement would be to include a more directed approach to waterbody selection, almost inevitably with waterbody types and sampling areas pre-determined. A similarly structured approach already underpins a range of existing programmes, including a number of FreshWater Watch projects (Lind et al., in this issue). For example, Florida's 'Lakewatch' volunteer water quality monitoring programme uses trained volunteers to collect scientifically robust total phosphorus, total nitrogen, chlorophyll *a*, and water clarity data from a large number of Florida lakes (Hoyer et al., 2012). Obrecht et al. (1998) used volunteers to monitor lake trophic state in Missouri, and found it reliable, and a range of volunteer river monitoring networks have been developed to augment the water quality data traditionally generated by professional monitoring (Loperfido et al., 2010).

Currently, however, just like statutory monitoring programs, citizen science water-quality monitoring projects consistently focus on larger waterbodies. Given the findings of the current survey, that some of the highest water quality is found in the smallest waterbodies, there is a strong rationale for encouraging freshwater citizen science monitoring programmes to cast the net wider.

Acknowledgements

We would like to thank Earthwatch and HSBC Bank who funded this study and provided all data for the citizen scientist element of the project under the scope of the FreshWater Watch, HSBC Water Programme. We are enormously grateful to the efforts of the citizen scientists who were active in gathering data for the project and Charlotte Hall who performed much of the training and feedback to keep the participants engaged. Thanks too, to Luis Felipe Velasquez, Eva Pintado Castilla and Andrew Henry for their assistance in GIS development and use, to Katy Williams and Steven Loiselle for editing support and to three anonymous reviewers for their valuable suggestions for improvements to the text.

References

Allan, J.D., 2004. Landscapes and riverscapes: the influence of land use on stream ecosystems. Annu. Rev. Ecol. Evol. Syst. 35, 257–284.

Baron, J.S., Poff, N.L., Angermeier, P.L., Dahm, C.N., Gleick, P.H., Hairston Jr., N.G., Jackson, R.B., Johnston, C.A., Richter, B.D., Steinman, A.D., 2003. Sustaining healthy freshwater ecosystems. Issues in Ecology 10:1–16 (Ecological Society of America. Available at: https://cfpub.epa.gov/watertrain/pdf/issue10.pdf).

Beaulac, M.N., Reckhow, K.H., 1982. An examination of land use – nutrient export relationships. J. Am. Water Resour. Assoc. 18, 1013–1024.

Berti, G., Fossati, P., Tarenghi, G., Musitelli, C., Melzi d'Eril, G.V., 1988. Enzymatic colorimetric method for the determination of inorganic phosphorus in serum and urine. Clin. Chem. Lab. Med. 26, 399–404.

Biggs, J., Williams, P., Whitfield, M., Nicolet, P., Weatherby, A., 2005. 15 years of pond assessment in Britain: results and lessons learned from the work of Pond Conservation. Aquat. Conserv. Mar. Freshwat. Ecosyst. 15, 693–714.

Biggs, J., Nicolet, P., Mlinaric, M., Lalann, T., 2014. Report of the Workshop on the Protection and Management of Small Water Bodies, Brussels, 14th November 2013. The European Environmental Bureau (EEB) and the Freshwater Habitats Trust (Report available at: http://www.eeb.org/EEB/?LinkServID=52A9ED8B-5056-B741-DBB0D3CCEA34187D).

Biggs, J., McGoff, E., Ewald, N., Dunn, F., Nicolet, P., Williams, P., 2016a. Clean Water for Wildlife. Using PackTest Nitrate and Phosphate Test Kits to Find Clean Water and Assess the Extent of Pollution. Freshwater Habitats Trust, Oxford.

Biggs, J., von Fumetti, S., Kelly-Quinn, M., 2016b. The importance of small waterbodies for biodiversity and ecosystem services: implications for policy makers. Hydrobiologia http://dx.doi.org/10.1007/s10750-016-3007-0 (in press).

Booth, D.B., Jackson, C.R., 1997. Urbanization of aquatic systems: degradation thresholds, stormwater detection, and the limits of mitigation. J. Am. Water Resour. Assoc. 33, 1077–1090.

Booth, D.B., Karr, J.R., Schauman, S., Konrad, C.P., Morley, S.A., Larson, M.G., Burges, S.J., 2004. Reviving urban streams: land use, hydrology, biology, and human behavior. American Water Resources Association 40, 1351–1364.

Boyer, T., Polasky, S., 2004. Valuing urban wetlands: a review of non-market valuation studies. Wetlands 24, 744–755.

Bradley, A., Frost, P.C. 2016 Monitoring water quality in Toronto's urban stormwater ponds: assessing participation rates and data quality of water sampling by citizen scientists in the FreshWater Watch (in this issue).

Brahney, J., Mahowald, N., Ward, D.S., Ballantyne, A.P., Neff, J.C., 2015. Is atmospheric phosphorus pollution altering global alpine lake stoichiometry? Glob. Biogeochem. Cycles 29, 1369–1383.

Briers, R.A., 2014. Invertebrate communities and environmental conditions in a series of urban drainage ponds in eastern Scotland: implications for biodiversity and conservation value of SUDS. Clean – Soil, Air, Water 42, 193–200.

Brown, C.D., Turner, N.L., Hollis, J.M., Bellamy, P.H., Biggs, J., Williams, P.J., Arnold, D.J., Pepper, T., Maund, S.J., 2006. Morphological and physico-chemical properties of British aquatic habitats potentially exposed to pesticides. Agric. Ecosyst. Environ. 113, 307–319.

Buytaert, W., Zulkafli, Z., Grainger, S., Acosta, L., Alemie, T.C., Bastiaensen, J., De Bievre, B., Bhusal, J., Clark, J., Dewulf, A., Foggin, M., 2014. Citizen science in hydrology and water resources: opportunities for knowledge generation, ecosystem service management, and sustainable development. Front. Earth Sci. http://dx.doi.org/10.3389/feart.2014.00026.

Canfield Jr., D.E., Brown, C.D., Bachmann, R.W., Hoyer, M.V., 2002. Volunteer Lake Monitoring: testing the reliability of data collected by the Florida LAKEWATCH Program. Lake and Reservoir Management 18, 1–9.

Carvalho, L., Phillips, G., Maberly, S.C., Clarke, R., 2006. UK Technical Advisory Group on the Water Framework Directive. UK environmental standards and conditions (phase 2). Final report. Available at:. http://nora.nerc.ac.uk/3305/1/WFD38_WP2_Final_report_e_version.pdf.

Cline, S.A., Collins, A.R., 2003. Watershed associations in West Virginia: their impact on environmental protection. J. Environ. Manag. 67, 373–383.

Davies, B.R., Biggs, J., Williams, P.J., Lee, J.T., Thompson, S., 2008a. A comparison of the catchment sizes of rivers, streams, ponds, ditches and lakes: implications for protecting aquatic biodiversity in an agricultural landscape. Hydrobiologia 597, 7–17.

Davies, B.R., Biggs, J., Williams, P., Whitfield, M., Nicolet, P., Sear, D., Bray, S., Maund, S., 2008b. Comparative biodiversity of aquatic habitats in the European agricultural landscape. Agric. Ecosyst. Environ. 125, 1–8.

Dias, N., Curwell, S., Bichard, E., 2014. The current approach of urban design, its implications for sustainable urban development. Procedia Economics and Finance 18, 497–504.

Dodds, W.K., Jones, J.R., Welch, E.B., 1998. Suggested classification of stream trophic state: distributions of temperate stream types by chlorophyll, total nitrogen, and phosphorus. Water Res. 32, 1455–1462.

Donohue, I., Styles, D., Coxon, C., Irvine, K., 2005. Importance of spatial and temporal patterns for assessment of risk of diffuse nutrient emissions to surface water. J. Hydrol. 304, 183–192.

Donohue, I., McGarrigle, M.L., Mills, P., 2006. Linking catchment characteristics and water chemistry with the ecological status of Irish rivers. Water Res. 40, 91–98.

Eigenbrod, F., 2016. Redefining landscape structure for ecosystem services. Current Landscape Ecology Reports 1, 80–86.

Gagnéa, S.A., Eigenbrod, F., Bert, D.G., Cunningtonc, G.M., Olsond, L.T., Smithe, A.C., Fahrig, L., 2015. A simple landscape design framework for biodiversity conservation. Landsc. Urban Plan. 136, 13–27.

Gerken Golay, M.E., Thompson, J.R., Mabry, C.M., Kolka, R.K., 2013. An investigation of water nutrient levels associated with forest vegetation in highly altered landscapes. Soil and Water Conservation Society 68, 361–371.

Gledhill, D.G., James, J., Davies, D.H., 2008. Pond density as a determinant of aquatic species richness in an urban landscape. Landsc. Ecol. 23, 1219–1230.

González Sagrario, M., Jeppesen, E., Gomà, J., Søndergaard, M., Jensen, J.P., Lauridsen, T., Landkildehus, F., 2005. Does high nitrogen loading prevent clear-water conditions in shallow lakes at moderately high phosphorus concentrations? Freshw. Biol. 50, 27–41.

Hassall, C., 2014. The ecology and biodiversity of urban ponds. WIREs Water 1, 187–206.

Hassall, C., Anderson, S., 2015. Stormwater ponds can contain comparable biodiversity to unmanaged wetlands in urban areas. Hydrobiologia 745, 137–149.

Heathwaite, A.L., Fraser, A.I., Johnes, P.J., Hutchins, M., Lord, E., Butterfield, D., 2003. The Phosphorus Indicators Tool: a simple model of diffuse P loss from agricultural land to water. Soil Use Mgt. 19, 1–11.

Hering, D., Aroviita, J., Baattrup-Pedersen, A., Brabec, K., Buijse, T., Ecke, F., Friberg, N., Gielczewski, M., Januschke, K., Köhler, J., Kupilas, B., Lorenz, A.W., Muhar, S., Paillex, A., Poppe, M., Schmidt, T., Schmutz, S., Vermaat, J., Verdonschot, P.F.M., Verdonschot, R.C.M., Wolter, C., Kail, J., 2015. Contrasting the roles of section length and instream habitat enhancement for river restoration success: a field study on 20 European restoration projects. J. Appl. Ecol. 52, 1518–1527.

Heron, J., 1961. The seasonal variation of phosphate, silicate, and nitrate in waters of the English Lake District. Limnol. Oceanogr. 6, 338–346.

Hill, M.J., Biggs, J., Thornhill, I., Briers, R.A., Gledhill, D.G., White, J.C., Wood, P.J., Hassall, C., 2016. Urban ponds as an aquatic biodiversity resource in modified landscapes. Glob. Chang. Biol. http://dx.doi.org/10.1111/gcb.13401.

Hoyer, M.V., Wellendorf, N., Frydenborg, R., Bartlett, B., Canfield, D.E., 2012. A comparison between professionally (Florida Department of Environmental Protection) and volunteer (Florida LAKEWATCH) collected trophic state chemistry data in Florida. Lake and Reservoir Management 28, 277–281.

James, C., Fisher, J., Russell, V., Collings, S., Moss, B., 2005. Nitrate availability and hydrophyte species richness in shallow lakes. Freshw. Biol. 50, 1049–1063.

John, B., Withycombe, L., Wieka, K.A., Langa, D.J., 2015. How much sustainability substance is in urban visions? – an analysis of visioning projects in urban planning. Cities 48, 86–98.

Johnes, P., Moss, B., Phillips, G., 1996. The determination of total nitrogen and total phosphorus concentrations in freshwaters from land use, stock headage and population data: testing of a model for use in conservation and water quality management. Freshw. Biol. 36, 451–473.

Karaus, U., Larsten, S., Guillong, H., Tockner, K., 2013. The contribution of lateral aquatic habitats to insect diversity along river corridors in the Alps. Landsc. Ecol. 28, 1755–1767.

Koperski, P., 2010. Urban environments as habitats for rare aquatic species: the case of leeches (Euhirudinea, Clitellata) in Warsaw freshwaters. Limnologica 40, 233–240.

Lambert, S.J., Davy, A.J., 2011. Water quality as a threat to aquatic plants: discriminating between the effects of nitrate, phosphate, boron and heavy metals on charophytes. New Phytol. 189, 1051–1059.

Latimer, J.S., Quinn, J.G., 1998. Aliphatic petroleum and biogenic hydrocarbons entering Narragansett Bay from tributaries under dry weather conditions. Estuaries 21, 91–107.

Lenat, D.R., Crawford, J.K., 1994. Effects of land use on water quality and aquatic biota of three North Carolina Piedmont streams. Hydrobiologia 294, 185–199.

Lind, K., Thornhill, I., Loiselle, S.A., 2016 A Cox regression model for relative engagement time and its application to participation in Citizen Science project (in this issue).

Liu, C., Kroeze, C., Hoekstra, A.Y., Gerbens-Leenes, W., 2012. Past and future trends in grey water footprints of anthropogenic nitrogen and phosphorus inputs to major world rivers. Ecol. Indic. 18, 42–49.

Loperfido, J.V., Beyer, P., Just, C.L., Schnoor, J.L., 2010. Uses and biases of volunteer water quality data. Environ. Sci. Technol. 44, 7193–7199.

Lottig, N.R., Wagner, T., Henry, E.N., Cheruvelil, K.S., Webster, K.E., Downing, J.A., Stow, C.A., 2014. Long-term citizen-collected data reveal geographical patterns and temporal trends in lake water clarity. PLoS One 9:e95769. http://dx.doi.org/10.1371/journal.pone.0095769.

Lubell, M., Schneider, M., Scholz, J.T., Mete, M., 2002. Watershed partnerships and the emergence of collective action institutions. Am. J. Polit. Sci. 46, 148–163.

Maas, R.P., Kucken, D.J., Gregutt, P.F., 1991. Developing a rigorous water quality database through a volunteer monitoring network. Lake and Reservoir Management 7, 123–126.

Martinez-Sanz, C., Cenzano, C.S.S., Fernandez-Alaez, M.F., Garcia-Criado, F., 2012. Relative contribution of small mountain ponds to regional richness of littoral macroinvertebrates and the implications for conservation. Aquat. Conserv. Mar. Freshwat. Ecosyst. 22, 155–164.

McKinney, M.L., 2008. Effects of urbanization on species richness: a review of plants and animals. Urban Ecosystems 11, 161–176.

Mekonnen, M.M., Hoekstra, A.Y., 2015. Global gray water footprint and water pollution levels related to anthropogenic nitrogen loads to fresh water. Environ. Sci. Technol. 49, 12860–12868.

Meybeck, M., 1998. Man and river interface: multiple impacts on water and particulates chemistry illustrated in the Seine River Basin. Hydrobiologia 373 (374), 1–20.

Meyer, L., Wallace, B., 2001. Lost linkages and lotic ecology: rediscovering small streams. 2000. In: Press, M.C., Huntley, N.J., Levin, S. (Eds.), Ecology: Achievements and Challenge. Blackwell Science, Oxford, UK, pp. 295–317.

Miller, S.W., Budy, P., Schmidt, J.C., 2010. Quantifying macroinvertebrate response to instream habitat restoration: applications of meta-analysis to river restoration. Restor. Ecol. 18, 8–19.

Moore, A.A., Palmer, M.A., 2005. Invertebrate biodiversity in agricultural and urban headwater streams: implications for conservation and management. Ecol. Appl. 15, 1169–1177.

Morse, C.C., Huryn, A.D., Cronan, C., 2003. Impervious surface area as a predictor of the effects of urbanization on stream insect communities in Maine, USA. Environ. Monit. Assess. 89, 95–127.
Morton, D., Rowland, C., Wood, C., Meek, L., Marston, C., Smith, G., Wadsworth, R., Simpson, I.C., 2011. Final Report for LCM2007 - The New Land Cover Map. CS Technical Report No 11/07. NERC/Centre for Ecology and Hydrology.
Moss, B., Jeppesen, E., Søndergaard, M., Lauridsen, T.L., Liu, Z., 2013. Nitrogen, macrophytes, shallow lakes and nutrient limitation: resolution of a current controversy? Hydrobiologia 710, 3–21.
Muenich, R.L., Peel, S., Bowling, L.C., Heller Haas, M., Turco, R.F., Frankenberger, J.R., Chaubey, I., 2016. The Wabash sampling blitz: a study on the effectiveness of citizen science. Citizen Science: Theory and Practice 1:1–15. http://dx.doi.org/10.5334/cstp.1.
Muneoka, T., Yamazaki, Y., Wakou, S., Kimura, M., 2014. Evaluation of nitrate pollution in river water at agricultural watershed. International Journal of Environmental and Rural Development 5, 51–56.
Nerbonne, J.F., Ward, B., Ollila, A., Williams, M., Vondracek, B., 2008. Effect of sampling protocol and volunteer bias when sampling for macroinvertebrates. J. N. Am. Benthol. Soc. 27, 640–646.
Nicholson, E.J., Ryan, J., Hodgkins, D., 2002. Community data-where does the value lie? Assessing confidence limits of community collected water quality data. Water Sci. Technol. 45, 193–200.
Obrecht, D.V., Milanick, M., Perkins, B.D., Ready, D., Jones, J.R., 1998. Evaluation of data generated from lake samples collected by volunteers. Lake and Reservoir Management 14, 21–27.
Ourso, R.T., Frenzel, S.A., 2003. Identification of linear and threshold responses in streams along a gradient of urbanization in Anchorage, Alaska. Hydrobiologia 501, 117–131.
Overdevest, C., Orr, C.H., Stepenuck, K., 2004. Volunteer stream monitoring and local participation in natural resource issues. Research in Human Ecology 11, 177–185.
Palmer, M.A., Hondula, K.L., Koch, B.J., 2014. Ecological restoration of streams and rivers: shifting strategies and shifting goals. Annu. Rev. Ecol. Evol. Syst. 45, 247–269.
Paul, M.J., Meyer, J.L., 2001. Streams in the urban landscape. Annu. Rev. Ecol. Syst. 32, 333–365.
Peckenham, J.M., Peckenham, S.K., 2014. Assessment of quality for middle level and high school student generated water quality data. J. Am. Water Resour. Assoc. 50, 1477–1487.
Phillips, G., Bennion, H., Perrow, M.R., Sayer, C.D., Spears, B.M., Willby, N., 2015. A review of lake restoration practices and their performance in the Broads National Park, 1980–2013. Report for Broads Authority, Norwich and Natural England. Available at:. http://www.broads-authority.gov.uk/__data/assets/pdf_file/0006/549114/Broads-Lake-Review.pdf.
Roach, W.J., Heffernan, J.B., Grimm, N.B., Arrowsmith, J.R., Eisinger, C., Rychener, T., 2008. Unintended consequences of urbanization for aquatic ecosystems: a case study from the Arizona Desert. Bioscience 58, 715–727.
Rotman, D., Preece, J., Hammock, J., Procita, K., Hansen, D., Parr, C., Lewis, D., Jacobs, D., 2012. Dynamic changes in motivation in collaborative citizen-science projects. Proceedings of the ACM 2012 Conference on Computer Supported Cooperative Work, pp. 217–226.
Roy, A.H., Rosemond, A.D., Paul, M.J., Leigh, D.S., Wallace, J.B., 2003. Stream macroinvertebrate response to catchment urbanisation (Georgia, USA). Freshw. Biol. 48, 329–346.
Seto, K.C., Güneralp, B., Hutyra, L.R., 2012. Global forecasts of urban expansion to 2030 and direct impacts on biodiversity and carbon pools. PNAS 109, 16083–16088.
Smith, R.F., Lamp, W.O., 2008. Comparison of insect communities between adjacent headwater and main-stem streams in urban and rural watersheds. J. N. Am. Benthol. Soc. 27, 161–175.
Sonoda, K., Yeakley, J.A., Walker, C.E., 2001. Near-stream landuse effects on streamwater nutrient distribution in an urbanizing watershed. J. Am. Water Resour. Assoc. 93, 144–154.
Steele, M.K., Heffernan, J.B., 2014. Morphological characteristics of urban water bodies: mechanisms of change and implications for ecosystem function. Ecol. Appl. 24, 1070–1084.
Thornhill, I., Loiselle, S.A., Lind, K., Ophof, D., 2016. The citizen science opportunity for researchers and agencies. Bioscience http://dx.doi.org/10.1093/biosci/biw089 (2016).
UKTAG, 2013. Updated recommendations on phosphorus standards for rivers. UK Technical Advisory Group on the Water Framework Directive. Available at:. http://www.wfduk.org/sites/default/files/Media/UKTAG%20Phosphorus%20Standards%20for%20Rivers_Final%20130906_0.pdf.
US EPA, 2015. Connectivity of Streams and Wetlands to Downstream Waters: A Review and Synthesis of the Scientific Evidence. Office of Research and Development, U.S. Environmental Protection Agency.
US Geological Survey, 1999. The quality of our nation's waters - nutrients and pesticides. USGS Circular 1225.
Verdonschot, R.C.M., Keizer-vlek, H.E., Verdonschot, P.F.M., 2011. Biodiversity value of agricultural drainage ditches: a comparative analysis of the aquatic invertebrate fauna of ditches and small lakes. Aquatic Conservation-Marine and Freshwater Ecosystems 21, 715–727.
Verdonschot, R.C.M., Kail, J., McKie, B.G., Verdonschot, P.F.M., 2016. The role of benthic microhabitats in determining the effects of hydromorphological river restoration on macroinvertebrates. Hydrobiologia 769, 55–66.
Völker, S., Kistemann, T., 2015. Developing the urban blue: comparative health responses to blue and green urban open spaces in Germany. Health & Place 35, 196–205.
Wang, L.Z., Lyons, J., Kanehl, P., Gatti, R., 1997. Influences of watershed land use on habitat quality and biotic integrity in Wisconsin streams. Fisheries 22, 6–12.
Williams, P., Whitfield, M., Biggs, J., Bray, S., Fox, G., Nicolet, P., Sear, D., 2004. Comparative biodiversity of rivers, streams, ditches and ponds in an agricultural landscape in Southern England. Biol. Conserv. 115, 329–341.
Williams, P., Biggs, J., Nicolet, P., 2010. Comment: new clean-water ponds—a way to protect freshwater biodiversity. British Wildlife 22, 77–85.
Wood, P.J., Greenwood, M.T., Agnew, M.D., 2003. Pond biodiversity and habitat loss in the UK. Area 35, 206–216.
Zhang, Y., Ma, R., Luo, J., Hu, M., Li, J., Liang, Q. 2016 Combining citizen science and land use data to identify drivers of eutrophication in the Huangpu River system (in this issue).