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CONSTRUCTED WETLANDS

Can an Integrated Constructed Wetland in Norfolk Reduce Nutrient Concentrations and Promote In Situ Bird Species Richness?

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Abstract

Integrated Constructed Wetlands (ICWs) are potentially effective tools in the effort to restore aquatic ecosystems, and also they incorporate multiple co-benefits. An ICW was constructed in Norfolk, UK, to address the degradation of a stream and lake receiving treated effluent from a small Sewage Treatment Works (STW). Results demonstrated that: (1) nutrient concentrations significantly reduced from the ICW influent to the effluent (percentage reductions: total phosphorus [TP]: 78%, orthophosphate: 80%, total oxidised nitrogen [TON]: 65%, nitrate: 65%, nitrite: 67%, ammoniacal nitrogen: 62%), and mean dissolved oxygen concentrations increased (influent mean: 6.4 ± 1.4 mg l^{-1} effluent mean: 17.8 ± 3.3 mg l^{-1}), (2) there were non-significant reductions in nutrient concentrations in the receiving stream (percentage reductions: TP: 23%, orthophosphate: 23%, TON: 26%, nitrate: 26%), with the exception of ammoniacal nitrogen (127% increase) and nitrite (76%) after ICW commissioning, and (3) mean in situ avian species richness increased from 10 to 27 species. Thus, the ICW substantially reduced nutrient concentrations, and had in situ conservation benefits. It is recommended that appropriately designed ICWs should be implemented widely and statutory authorities should ensure: 1) best-practice maintenance and 2) final effluent monitoring at both the STW and at the ICW outflows.

Keywords Phosphorus · Water quality · Biodiversity · Birds · Land treatment area · Environmental permits

Introduction

Despite improvements in the treatment of domestic and industrial wastewaters, the current levels of nutrient fluxes in sewage effluent remain substantially higher than at the beginning of the twentieth century, and are frequently far higher than would be required to elicit ecological recovery of surface waters (Sayer et al. 2010; Bowes et al. 2012, 2016; Naden et al. 2016; McCall et al. 2017). Small rural sewage treatment works (STWs) have been implicated in the eutrophication of surface waters in the United Kingdom (Mainstone and Parr 2002; Neal et al. 2005; Jarvie et al. 2006; Roberts and Cooper 2018). Elevated nutrient concentrations in receiving waters downstream of STWs have been shown to result in excessive phytoplankton and periphyton growth which can compromise both the ecology and the ecosystem services provided by waterbodies (Jeppesen et al. 2000; Flynn et al. 2002; Hutchins et al. 2010; Bowes et al. 2012).

The county of Norfolk lies in the east of England. The county's network of rivers and lakes has experienced impacts from diffuse and point source inputs of nutrients over many decades (Moss 1983, 2001; Sayer et al. 2010). A recent assessment undertaken by the government regulator in England (Environmental Agency 2016) reported that 89% of region's 603 waterbodies failed to achieve good ecological status under the Water Framework Directive (WFD) of the European Union (EC 2000). Point sources of phosphorus (P) from STWs have been implicated as contributory factors to the progressive degradation of waterbodies across Norfolk (Robson and Neal 1997; Lau and Lane 2002; Demars et al.

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2005; Phillips et al. 2005) and they contribute to the failure to achieve good ecological status under WFD for at least 50% of the waterbodies in the region (Environmental Agency 2016).

Phosphorus fractions are commonly the limiting nutrients in freshwaters, and so reducing P concentrations is central to combatting eutrophication (Moss 2001; Jarvie et al. 2002, 2018). A study of 76 Danish shallow lakes suggested that a reduction in the species richness of submerged aquatic plants and other functional and taxonomic groups occurs above total phosphorus (TP) concentrations of around 0.050 mg l^{-1} (Jeppesen et al. 2000). Proposed water quality objectives to restore lake communities in the Broads National Park in Norfolk would require (TP) concentrations of ≤0.040 mg l^{-1}, although it is acknowledged that 0.060 mg l^{-1} is likely to be more realistic (Phillips et al. 2015). In rivers, the more relevant fraction of P is believed to be soluble reactive phosphorus (SRP) (Mainstone and Parr 2002; Hilton et al. 2006; Jarvie et al. 2006). In this work, SRP will be used synonymously with orthophosphate, although it is acknowledged that the methodology for determination of "orthophosphate as P" by the National Laboratory Service (NLS), is slightly different from that for "true" orthophosphate/SRP as described by Jarvie et al. (2002). Pragmatic water quality targets for river systems are in the range 0.020–0.100 mg l^{-1} of orthophosphate with an intermediate target for heavily enriched rivers of 0.200 mg l^{-1} (Mainstone and Parr 2002). However, more recent evidence shows that concentrations may need to be lower if the ecological community is to respond as demonstrated by mesocosm experiments showing that periphyton communities were only limited by orthophosphate below around 0.033 mg l^{-1} in a chalk stream (McCall et al. 2017), with similar results reported for other river typologies (McCall et al. 2014). Moreover, even significant reductions in inputs may be buffered by legacy sources of P from the catchment and aquatic sediments which can persist from months to centuries (Jarvie et al. 2013; Sharpley et al. 2014). Consequently, a holistic and long term view which encompasses multiple benefits of schemes is important when P management measures are considered (Scholz et al. 2007).

There is a high degree of variability within the literature regarding the removal efficiencies of nutrients by wetlands. Natural wetlands have been reported to remove high proportions of influent nutrients, but may also act as sources of N and P some or all of the time (Fisher and Acreman 2004; Vymazal 2007). The concept of using constructed wetlands to moderate pollution and treat wastewater is well established (Kadlec and Knight 1996; Mander and Mitsch 2009; Vymazal 2009) and it has been estimated that more than 1200 constructed wetlands are currently in operation in the UK (Cooper 2009). However, many of these systems follow a formulaic, engineering approach (Cooper et al. 1996) and operate within a consensual paradigm derived from the early twentieth century, rather than one that embraces the potential to deliver multiple and systemic solutions (Everard and McInnes 2013). Whilst narrowly framed solutions can make a positive contribution to water quality management issues, often the wider socio-ecological linkages and the opportunities to optimise co-benefits remain unfulfilled (Chan et al. 2006; Everard et al. 2012). The integrated constructed wetland (ICW) concept is a multifunctional, multi-benefit approach that has at its core the tripartite aims of water quantity and quality management, biodiversity enhancement and landscape fit (Harrington and McInnes 2009; Everard et al. 2012). ICWs are also characterised by a range of specific yet flexible design specifications (Carty et al. 2008). Evidence is available which relates features such as geometry and size to water treatment efficiency (Scholz et al. 2007), and optimisation for biodiversity (Becerra-Jurado et al. 2012). The efficacy of ICWs to deliver multiple benefits has been widely tested and documented (for instance in Scholz et al. 2007; Babatunde et al. 2007; Harrington and McInnes 2009; Mustafa et al. 2009; Kayranli et al. 2010; Zhang et al. 2008). Research on ICWs treating both domestic wastewater and agricultural point-source pollution has demonstrated their ability to sustainably reduce nutrient concentrations (Doody et al. 2009; Mustafa et al. 2009; Dong et al. 2011) and that well-designed ICWs can be the most effective form of constructed wetland for wastewater treatment (Hickey et al. 2018). Orthophosphate reduction rates between influent and effluent concentrations over a 9 year period from twelve ICWs in the Annestown Stream catchment, County Waterford, Ireland ranged between 81.36% and 99.71% and ammonium-N removal rates ranged between 95.26% and 99.34% (Harrington and McInnes 2009).

Constructed wetlands have the potential to not only improve water quality but also to increase in situ biodiversity (Hsu et al. 2011; Semeraro et al. 2015). Becerra Jurado et al. (2010) demonstrated that an ICW can support the same macroinvertebrate diversity of taxa as natural ponds. In a further study on Coleoptera, Becerra-Jurado et al. (2014) demonstrated that the various cells of an ICW supported 82 water beetle taxa, which represents 26% of the known Irish aquatic lentic coleopteran fauna and included several species of conservation concern including the 'endangered' species *Agabus conspersus* and *Berosus signaticollis*. When compared to artificial wetlands, natural wetlands support a greater abundance and diversity of birds (Ma et al. 2004; Brooks et al. 2005; Hsu et al. 2011), but artificial and managed wetlands can still offer an important habitat for a range of bird species (Sebastián-González et al. 2010; Lewis-Phillips et al. 2019) providing refuge, food and breeding sites (Sebastián-González et al. 2010). Increased macrophyte coverage and complexity supports greater invertebrate density (Declerck et al. 2011) providing a food source for some insectivorous bird species. However, potential negative outcomes such as exposing wildlife to pathogens or pollutants have been highlighted as a concern (Chen 2011). Moreover, while determinants of bird diversity in wetlands are well-documented, the tailoring of design specifications to optimise multiple goals may mean that it is not possible to maximise all outcomes. For example a focus on nutrient removal can result in reduced capacity for diversity and ecosystem functioning, and exceedance of critical nutrient loadings may also have a negative impact on biodiversity and ecosystem functioning (Richardson and Qian 1999; Hansson et al. 2005; Verhoeven et al. 2006).

In addition to enhancing biodiversity, ICWs also have the potential to deliver on range of ecosystem services which underpin the well-being of local stakeholders (Harrington and McInnes 2009). Evidence from the UK indicates that when the full range of benefits are embedded in the decision-making process and considered against traditional approaches to the management of P in wastewater, an ICW provides greatest utility and value for money (McInnes et al. 2016).

This paper provides an initial assessment of the ability of an ICW in Norfolk, UK, (henceforth referred to as Frogshall ICW) which was commissioned in the autumn of 2014, to improve water quality discharging from a small rural STW (Northrepps STW) into a headwater chalk stream and on-stream lake. Changes in avian diversity before and after ICW construction were also assessed. The main objectives were to (1) quantify water quality alterations occurring as water transits through the ICW (2) determine to what extent such alterations in water quality resulted in corresponding changes in downstream water quality, and (3) record changes in on-site avian biodiversity after ICW construction. The threshold concentrations used as reference points to determine whether water quality objectives could be met using the ICW were taken from the literature on similar systems, a chalk stream (0.033 mg l^{-1} orthophosphate from McCall et al. [2017]) and the nearby system of lakes called the Norfolk Broads (0.040 mg l^{-1} TP from Phillips et al. [2015]).

Methods

Study Site

The Frogshall ICW is located close to the village of Northrepps, and is shown in Fig. 1 (Norfolk, UK, latitude: 52° 53′ 40.34″ N, longitude: 001° 20′ 51.05″ E). The site was originally rough pasture and was developed for an ICW by Norfolk Rivers Trust (NRT) in October 2014, with full commissioning taking place by 1 November 2014. The primary purpose of constructing the ICW was to improve the water quality of the effluent after it had discharged from Northrepps STW (population equivalent: 553) in order to improve downstream water quality and allow the downstream ecosystem to recover from its degraded state. Prior to the construction of the ICW, the works discharged directly into the headwaters of the Mundesley Beck, a chalk stream, contributing 100% of the initial flows to the stream under most flow conditions. The Mundesley Beck in turn discharges into a shallow lake (Little Broad, latitude: 52° 53′ 31.66″ N, longitude: 001° 21′ 39.26″ E) 700 m downstream, and the lake had become severely eutrophic over time, normally manifest by over 50% cover of floating algae from April–August and little macrophyte growth. Improvements in water quality, especially reductions in phosphate, were deemed desirable to reduce eutrophication of the lake and suspected impacts on the stream ecosystem. The landowner of the lake and the field where the ICW was subsequently constructed was supportive of the project due to the nature conservation and wider benefits which the wetland could bring.

Prior to construction, archaeological, utility, soil hydrological and ecological searches and surveys were conducted. Prior to proposing the ICW, water quality monitoring had enabled an evaluation of the role of the STW in producing the high concentrations of P in the water course and Little Broad. The water company (Anglian Water) was then approached by NRT and agreed to let the Trust connect the final effluent from the works to the proposed ICW.

To connect the STW to the ICW, the final effluent from the STW was collected by a concrete control structure which was designed such that if the pipe become blocked effluent would overflow and could not back up and compromise the STW, whilst also not preventing water samples from being taken from the original outfall of the STW, as required for compliance purposes (Fig. 2). The STW effluent was then piped 180 m to the first cell in the ICW. The ICW construction consisted of three shallow ponds, or cells, (0.1–0.3 m water depth) with a total area of 2953 m^2 (0.3 ha). Levels and geometry were set such that hydrological short circuiting would not occur and spoil was used to landscape the edges of the ICW. Individual cell areas are shown in Fig. 1. Flow between cells and water levels were controlled by twin-wall 300 mm diameter plastic pipes with adjustable 90 degree elbow joints to allow water level control. Once the wetland cells had been constructed, they were planted with approximately 15,000 aquatic and emergent plants including the following species: *Iris pseudacorus, Carex riparia, Sparganium emersum, Equisetum fluviatile, Eleocharis palustris, Carex rostrate, Alsima plantago-aquatica, Mentha aquatic, Persicaria amphibia, Veronica beccabunga*. Growth was rapid, and the ICW rapidly progressed to a stage where it was dominated by emergent plants. Common reed *(Phragmites australis*) was avoided due to previous experience showing that it produces thick stands which can promote anaerobic conditions which may in turn contribute to P release from sediment (McInnes *pers comm.* 2014). The NRT were able to minimise the costs of construction by the agreement of the landowner not to charge for the land where the ICW was constructed, and community volunteers helped with the planting of aquatic

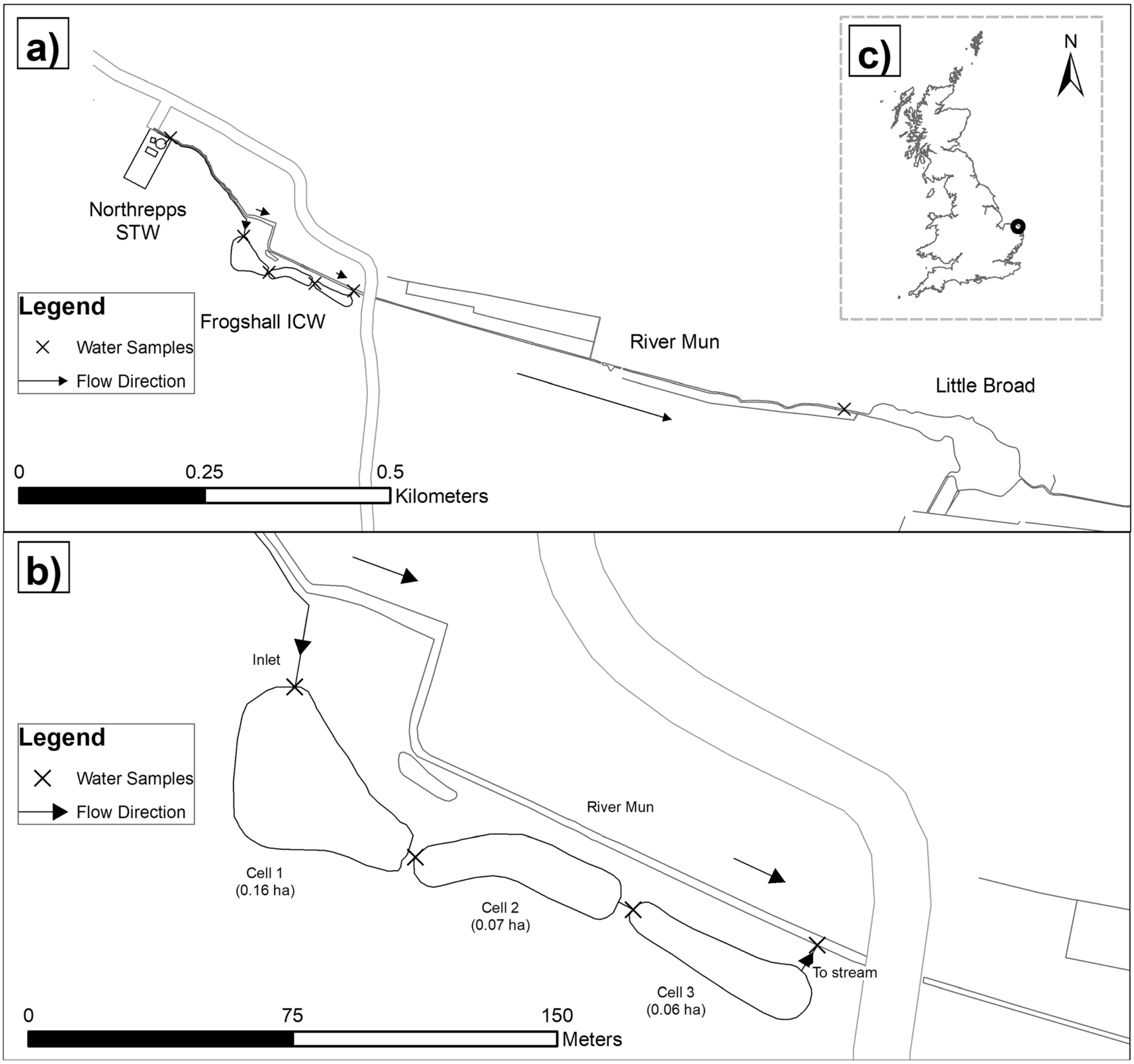

Fig. 1 Maps showing: a) the upper Mundesley Beck near Northrepps, Norfolk, UK, including Northrepps sewage treatment works (STW), the Frogshall integrated constructed wetland (ICW), water sampling locations and Little Broad b) Frogshall ICW, and c) the location of the study site within the UK for context. Maps were generated using OS Mastermap® in ArcGIS 10.5 (Ordnance Survey (GB) 2018)

plants. Capital costs are shown in Table 1. During the monitoring period 2014–2016 only minor maintenance occurred which comprised thinning out aquatic plants in the wetland cells to ensure some open water was maintained to attract aquatic birds.

Water Quality Monitoring and Acquisition of Hydrological Data

Water samples were taken at the nine surface water contributions to the Mundesley Beck upstream of Little Broad in order to quantify potential nutrient sources in December 2013 and February 2014. Based on this preliminary data, samples were collected on an approximately monthly basis at the locations which were deemed the most informative in diagnosing the source of nutrients, and with a view to gather baseline data preceding any intervention. The sampling points relevant to this study are shown in Fig. 1, sampling points which were not regularly re-sampled have not been shown in Fig. 1 to avoid confusion. Monthly water samples were collected from the Mundesley Beck downstream of the STW and upstream of the on-stream lake (Little Broad) in the months specified in Table 2. Once the ICW was constructed, monthly samples were additionally collected from the locations within the

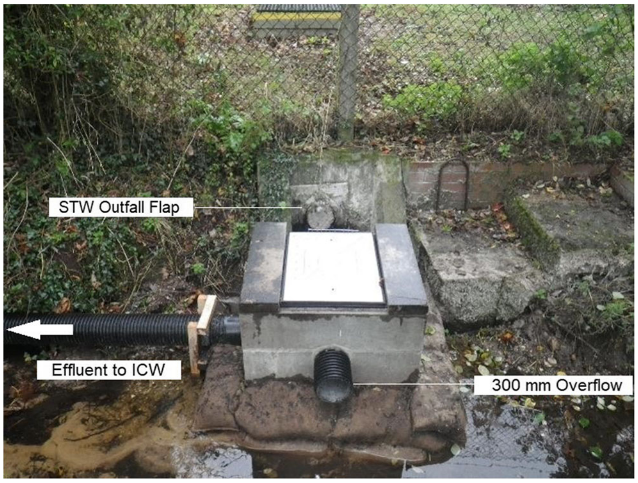

Fig. 2 Structure designed to collect effluent from sewage treatment works (STW) and convey effluent to Frogshall integrated constructed wetland (ICW). The upper invert of the both the overflow and conveyor pipes were lower that the bottom of the STW pipe invert

ICW treatment train from December 2014 – September 2016 (Table 2). Water samples were collected, unfiltered, in acid washed one litre polyethylene terephthalate (PET) plastic bottles supplied by the Environment Agency NLS. Samples were collected by courier on the day of sampling and conveyed to the NLS. Dissolved oxygen concentrations were measured monthly from April – June 2015 in the influent to the ICW and at the effluent from each wetland cell using a Hanna HI 9146 dissolved oxygen meter which was calibrated prior to each visit (Table 2).

The water company provided 9 months of daily flow data (m^3 d^{-1}) from the works for the period 1 October 2014–1 July 2015. These data were used to estimate the annual load of total phosphorus (TP) discharging from the STW by converting the mean daily discharge of 71 ± 4 m^3 d^{-1} (henceforth the symbol ± represents one standard deviation) into litres and then multiplying by the mean TP (mg l^{-1}; based on seven water samples) concentration for the period. In the absence of a full year of discharge data, the assumption was made that the 9 months of data (which covered discharges from all four seasons) were representative of the full year.

Table 1 Capital costs of Frogshall integrated constructed wetland ICW

Item	Cost Including VAT (£)
Site Works	
Surveys, applications for permissions	£1305
Earthworks, pipework and site supervision	£21,712
Wetland Plants	£7004
Monitoring	
Water quality analysis	£6900
Stream invertebrate analysis	£4000
Total	**£40,920**

Table 2 Water samples were collected from the sewage treatment works (STW) effluent, within the integrated constructed wetland (ICW) and in the downstream receiving watercourse (Mundesley Beck). All nutrient water samples were analysed for total phosphorus (TP), orthophosphate, total oxidised nitrogen (TON), nitrate, nitrite, and ammoniacal nitrogen. Dissolved oxygen observations were recorded within the ICW April – June 2015

Site	Data available	Dates collected	Number of observations or samples
ICW influent (STW effluent), Effluent from cells 1–3	Nutrients	December 2014 – June 2015, October 2015, December 2015, February 2016, July–September 2016	13
ICW influent (STW effluent), Effluent from cells 1–3	Dissolved oxygen	April – June 2015	3
Receiving watercourse 5 m downstream of ICW outfall	Nutrients	April–August 2014, March 2015, May 2015, June–July 2016	9
Receiving watercourse 700 m downstream of ICW outfall	Nutrients	December 2013 – February 2014, April–August 2014, December 2014–June 2015, December 2015, February 2016, July–August 2016	19

Whilst this assumption cannot be validated, visual appraisal of the flow data and the small standard deviation (SD) both suggest that the daily discharge from the works was relatively constant throughout the year.

In order to describe the hydrological context during baseline (December 2013 – August 2014) and post-commissioning (December 2014 – August 2016) periods, precipitation data were gathered from a weather station which was <14 km to the west of the site (Mannington Hall, Source ID: 24219), and temperature data were from accessed from the UK Meteorological Office's regional climate summaries for East Anglia (Met Office 2019a, b). These data were used to calculate the net precipitation (precipitation minus potential evapotranspiration [PET]), which has been used as an informative indicator of changes the hydrological conditions (for instance Thompson et al. 2009; Singh et al. 2010). PET was calculated according to Hargreaves PET, using the regional climate summary temperature records, as recommended by the Food and Agriculture Organisation of the United Nations (FAO) when there is insufficient data for calculation of Penman-Monteith PET (Hargreaves and Samani 1985; Allen et al. 1998; Hargreaves and Allen 2003).

Additionally to the data collected as part of this project, the Environment Agency supplied water quality data from a monitoring point approximately 4 km downstream of the ICW (Gimingham Mill Pool, latitude: 52° 52′ 51.36″ N, longitude: 001° 23′ 47.69″ E) which was collected approximately monthly from January 2000 until August 2018. This data constituted 130 samples in the baseline period and 16 samples after the ICW was commissioned. To assess potential hydrological influences on these data, UK Met Office regional climate summary temperature data and precipitation data from the Mannington Hall weather station were again used to determine annual net precipitation as previously described (Met Office 2019a, b).

Water Sample Analysis

Water samples were sent to the NLS for analysis on the day of sampling. Analysis was performed for total phosphorus as P (TP), orthophosphate as P, nitrate as N, nitrite as N, total oxidised nitrogen as N (TON), and ammoniacal nitrogen as N.

To determine TP, a persulphate digestion was first performed followed by the phosphomolybdenum blue colorimetry method at 880 nm using an AquaKem Discrete Analyser. Colorimetric analysis for orthophosphate, TON, nitrate, and ammoniacal nitrogen were carried out on filtered samples using a Konelab Discrete Analyser. Orthophosphate was determined following Murphy and Riley (1962) by complexation with ammonium molybdate and antimony potassium tartrate under acidic conditions, followed by colorimetric analysis at 880 nm. Nitrite was determined by diazotisation with sulphanilamide and coupling with N-(1-naphthyl)-ethylenediamine dihydrochloride. The coloured azo-dye absorbance was measured at 540 nm. To determine TON, nitrate within samples were first reduced to nitrite with hydrazine sulphate. Nitrite ions produced, together with those already present, were determined by the method for nitrite described above. Nitrate was determined by subtracting nitrite from TON. To determine ammoniacal nitrogen, salicylate and dichloroisocyanurate were added to samples in the presence of sodium nitroprusside thus forming a blue colouration proportional to the ammoniacal nitrogen concentration. Sodium citrate was added to mask any interference from other cations, and the colour produced was measured at 660 nm. The target relative standard deviation was <5% for all analyses. The accuracy of the data NLS produces is assessed in inter-lab calibration as prescribed by the UK Accreditation Service (UKAS 2019).

Statistical Methods (Water Quality)

Statistical tests carried out in IBM SPSS Statistics V22.0 Software determined whether there were significant differences between 1) mean nutrient concentrations in the influent to the ICW as opposed to the effluent and 2) mean nutrient concentrations in the Mundesley Beck before and after the ICW was commissioned. Data were first tested for normality. Where data were non-normal they were log transformed and re-tested for normality. After transformation, data were normally distributed. Log transformed data were used for statistical testing, but data which are presented are in raw form. Significant differences between influent and effluent nutrient concentrations were determined using Student's paired t-tests with a significance threshold at the $P = 0.05$ level. To ensure that data collected by NRT from the Mundesley Beck before and after ICW commissioning were seasonally comparable, only data from meteorological winter, spring, and summer were used in statistical comparisons. Autumn months were underrepresented in baseline data so data were excluded. Data supplied by the Environment Agency for the period 2000–2018 were sufficiently representative of all seasons so aggregated baseline data were simply compared to aggregated post-intervention data. Significant differences in river water chemistry before and after were determined using independent t-tests with a significance threshold at the $P = 0.05$ level.

Monitoring of Birds

Annual bird monitoring was conducted on the rough pasture field in May and June 2014 prior to the construction of the ICW, and for two successive years after ICW construction in 2015 and 2016. Surveys were conducted following the British Trust for Ornithology Breeding Bird Survey methodology protocol (BTO 2016), following a set 250 m transect route through the site. Surveys were conducted between 30 mins before sunrise and 10 am. Birds within the site were recorded, documenting how the individual or group were initially identified, divided into visual, call or song. The same surveyor completed all three surveys to ensure continuity of recordings.

Results

Water Quality

Mean concentrations of all nutrients were significantly lower in the effluent from the ICW as compared to the influent, with reductions ranging from 62 to 80% (Table 3). Concentrations of TP and TON from the STW, constituting the influent, varied relatively little from their mean values of 8.53 ± 0.77 mg P l^{-1} and 48.08 ± 5.98 mg N l^{-1} respectively. TP and TON in the effluent were relatively more variable at 1.89 ± 0.78 mg P l^{-1} and 16.96 ± 6.64 mg N l^{-1} respectively.

Nutrient concentrations reduced progressively though the ICW treatment train, with the greatest reductions in TP, orthophosphate, TON and nitrate occurring in cell one which represented 54% of the ICW area (Fig. 3). Figure 4 demonstrates

Table 3 Efficiency of Frogshall ICW during the period December 2014–September 2016 (number of samples; $n = 13$). Outcomes of paired t-tests for significant differences between ICW influent and effluent are indicated: * Significant ($P < 0.05$), ** highly significant ($P < 0.01$) *** very highly significant different ($P < 0.001$)

	n	Mean influent (mg l^{-1})	Influent SD (mg l^{-1})	Mean Effluent (mg l^{-1})	Effluent SD (mg l^{-1})	Change (%)
Total phosphorus	13	8.53	0.77	1.89	0.79	−78 ***
Orthophosphate	13	7.62	1.13	1.53	0.71	−80 ***
Total oxidised nitrogen	13	48.08	5.98	16.96	6.64	−65 ***
Nitrate	13	47.71	5.92	16.80	6.66	−65 ***
Nitrite	13	0.41	0.25	0.14	0.08	−67 ***
Ammoniacal nitrogen	13	0.28	0.32	0.10	0.15	−62 *

that there is a strong relationship between ICW area and the cumulative nutrient reductions in successive cells through the treatment train. The strength of the relationship appears to continue to be strong even in the ICW cells further downstream in the treatment train (Fig. 4). As previously described, impairment of ecosystems might be expected above approximately 0.033 mg l^{-1} of orthophosphate in chalk streams, and above 0.050 mg l^{-1} of TP in lakes (Phillips et al. 2015; McCall et al. 2017). Using the formulae shown in Fig. 4, relating ICW area to orthorphosphate and TP concentrations in the effluent, the total ICW area which would be required to drive concentrations below both these thresholds was estimated to be ≥1.1 ha, a 3.6-fold increase in ICW area from its current footprint.

Dissolved oxygen concentrations were measured monthly from April – June 2015, and mean values were 6.4 ± 1.4 mg l^{-1} in the influent, 17.8 ± 3.3 mg l^{-1} at the outflow of cell one, 18.1 ± 1.0 mg l^{-1}) at the outflow of cell two, and 16.9 ± 2.0 mgl^{-1} at the outflow to cell three. This pattern corresponds to a rapid increase in oxygen concentrations by the outflow of cell one, and a stabilisation in subsequent ponds.

Receiving Watercourse, the Mundesley Beck

The nine surface water inputs (in addition to the STW) joining the Mundesley Beck upstream of Little Broad were sampled on two separate occasions during the project scoping stage. Nutrient concentrations ranged from 3.55–15.70 mg N l^{-1} for TON, 3.54–15.70 mg N l^{-1} for nitrate, 0.00–0.08 mg N l^{-1} for nitrite, 0.01–0.04 mg N l^{-1} for ammoniacal nitrogen, 0.05–0.23 mg P l^{-1} for TP and 0.03–0.20 mg P l^{-1} for orthophosphate. Additionally, nutrients were sampled in the Mundesley Beck at intervals downstream from the ICW before and after it was

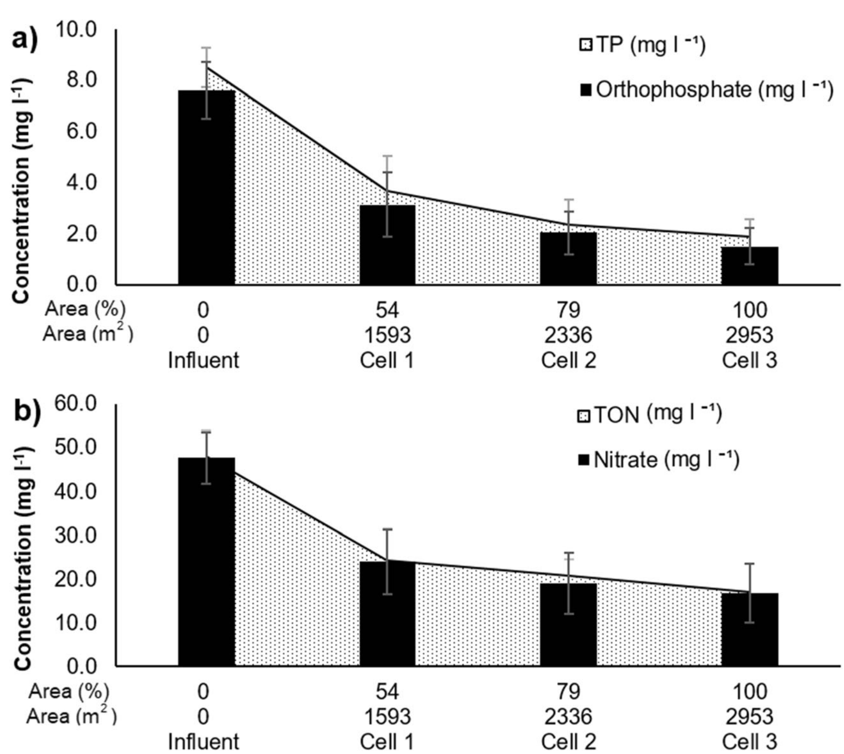

Fig. 3 Nutrient concentrations within the ICW treatment train, showing: a) mean concentrations of total phosphorus (TP) and orthophosphate from the influent of the Frogshall ICW through the three successive ICW cells, and b) concentrations of total oxidised nitrogen (TON) and nitrate from the influent of the ICW through the three successive ICW cells. The cumulative area is shown as a percentage of the overall total and in meters squared. The data represent mean values of 13 paired samples collected December 2014 – September 2016

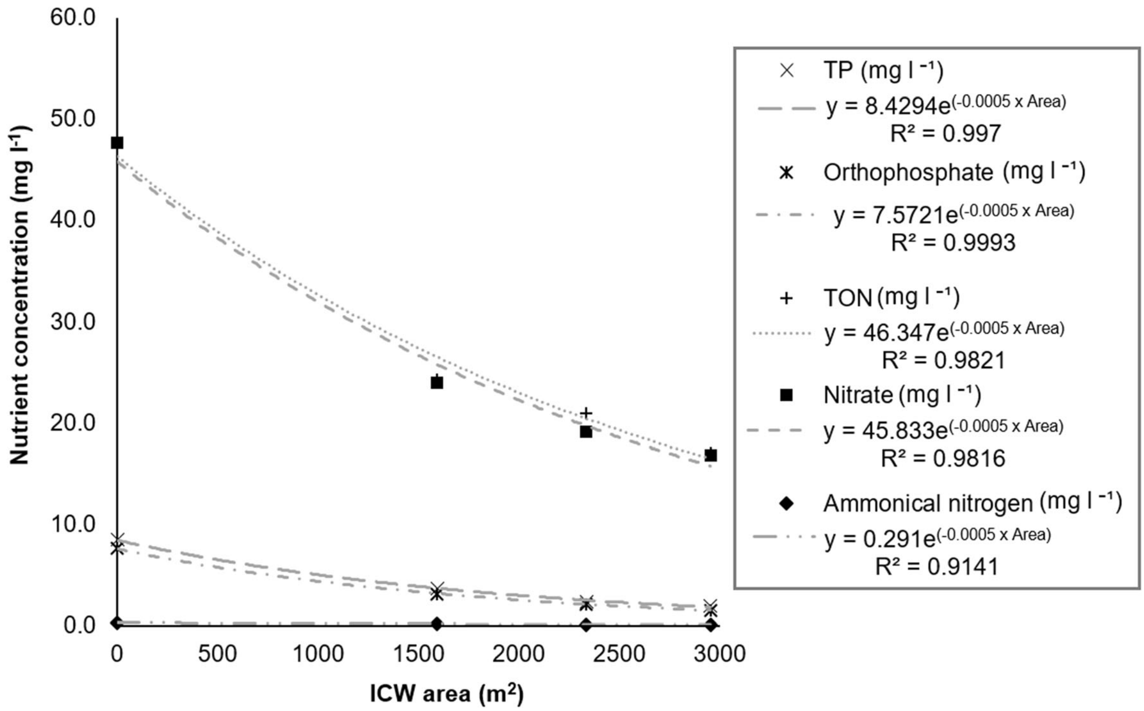

Fig. 4 Frogshall ICW area (m^2) plotted against mean nutrient concentrations from 13 sampling visits taken at the discharge from the influent, effluent and each cell (December 2014 – September 2016). Lines were fitted in Microsoft Excel (2013), and the exponential functions are shown in the legend

commissioned. The net precipitation shown in Table 4 demonstrate that conditions were wetter during the baseline than the post-commissioning period, suggesting that the post-commissioning stream discharges would have been less great thus causing less dilution. Over the longer time-period 2000–2018, relevant only to the receiving watercourse monitoring position 4 km downstream of the ICW, mean net precipitation before ICW commissioning was 47.3 ± 136.7 mm yr^{-1}, and for post-intervention was -34.3 ± 91.0 mm yr^{-1}. Thus, again the baseline period corresponding to this site was similar, but slightly wetter than the post-intervention period. Concentrations of all nutrients decreased after the ICW was commissioned as they had done within the ICW itself, apart from ammoniacal nitrogen and nitrite which increased (Table 5). Only the increase of nitrite 700 m downstream of the ICW was statistically significant at the $P = 0.05$ level.

Bird Species Richness

Ten species of bird were recorded on the site in 2014, prior to the installation of the ICW. Post-commissioning, species richness increased to 28 in 2015, with 26 species recorded in 2016. Using British Trust for Ornithology (BTO) conservation categories (BTO 2016), the number of red listed species on site increased from zero prior to installation to an mean of 4 species in the 2 years after wetland commissioning, and amber listed species increased from one to an average of 6.5 in the 2 years subsequent to wetland commissioning (Fig. 5; species list included in Online Resource 1).

Table 4 Hydrological conditions during the study period expressed as net precipitation (precipitation minus PET). Precipitation data were gathered from the nearby Mannington Hall weather station (Source ID: 24219; Met Office 2019b) and temperature data, used to calculate Hargreaves PET (Hargreaves and Samani 1985; Hargreaves and Allen 2003), were accessed from the UK Meteorological Office's climate summaries for East Anglia (Met Office 2019a). Negative values show that net evapotranspiration exceeds precipitation, as has previously been observed during the spring and summer in other analysis of climatological data from north Norfolk (Clilverd 2016)

Season	Net precipitation before commissioning (2013–14) (mm)	Net precipitation during commissioning (2014) (mm)	Net precipitation, year 1 post-commissioning (2014–15) (mm)	Net precipitation, year 2 post-commissioning (2015–16) (mm)
Winter	298.0		140.3	164.7
Spring	−25.6		−95.3	−49.0
Summer	−115.0		−141.4	−213.2
Autumn		70.2	97.3	87.8

Table 5 Changes in nutrient concentrations within the receiving watercourse (Mundesley Beck) before and after commissioning of Frogshall ICW. Independent t-tests were carried out for differences before and after the intervention, and the * symbol denotes significance at the $P = 0.05$ level

	Before ICW			After ICW			
	n	Mean	SD	n	Mean	SD	Change %
Receiving watercourse 5 m downstream of ICW discharge							
Total phosphorus	5	1.39	0.98	4	1.08	0.15	−23
Orthophosphate	5	1.23	0.85	4	0.94	0.25	−23
Total oxidised Nitrogen	5	17.21	5.78	4	12.71	2.90	−26
Nitrate	5	17.16	5.76	4	12.61	2.93	−26
Nitrite	5	0.06	0.03	4	0.10	0.07	76
Ammoniacal nitrogen	5	0.06	0.06	4	0.13	0.12	127
Receiving watercourse 700 m downstream of ICW discharge							
Total phosphorus	8	0.66	0.71	11	0.57	0.19	−14
Orthophosphate	8	0.55	0.57	11	0.50	0.18	−9
Total oxidised Nitrogen	8	15.46	2.25	11	14.62	1.70	−5
Nitrate	8	15.45	2.24	11	14.58	1.71	−6
Nitrite	8	0.03	0.02	11	0.05	0.03	71 *
Ammoniacal nitrogen	8	0.03	0.02	11	0.04	0.03	50
Receiving watercourse 4 km downstream of ICW discharge							
Orthophosphate	130	0.10	0.09	16	0.09	0.05	−14

Discussion

In Situ Water Quality

Between the influent and effluent of the ICW there were statistically significant reductions in the concentrations of TP and orthophosphate by 78% and 80% respectively (mean influent: 8.5 mg l^{-1}, mean effluent: 1.9 mg l^{-1}; and mean influent: 7.6 mg l^{-1}, mean effluent: 1.5 mg l^{-1} respectively). In a review of the efficiency of constructed wetlands, Vymazal (2007) reported that removal efficiencies of TP ranged from 42 to 60%, with Free Water Surface (FWS) wetlands in the middle of this range with a mean removal efficiency for TP of 49% (mean influent: 4.2 mg l^{-1}, mean effluent: 2.15 mg l^{-1}). Whilst TP and orthophosphate represent different P fractions and cannot be directly compared, this contrasts with efficiencies for the subset of FWS wetlands termed ICWs, have mean removal efficiencies for orthophosphate in the range 82% – 99.7% for 12 wetlands constructed to treat farm yard dirty water and one to treat domestic waste water (Harrington and McInnes 2009; Dong et al. 2011). Thus the Frogshall ICW removal efficiencies for TP and orthophosphate indicate that it operates towards the higher end of effectiveness for constructed wetlands as reported by Vymazal (2007), and at the lower end of the efficiency ICW efficiency range. Similarly, reductions in the concentrations of TON, nitrate and ammoniacal nitrogen, were similar to, or exceeded those found by Vymazal (2007), but were modest in comparison to other ICWs (Doody et al. 2009; Dong et al. 2011).

It is probable that the nutrient removal efficiencies for the ICW fall at the lower end of the ICW range because, given constraints imposed by the area of land available and cost, the size of the ICW was sub-optimal in comparison to the volume and nutrient concentrations to be treated. Harrington and McInnes (2009), found that functional ICW area was strongly related to P removal. The Frogshall ICW was fed by a STW discharging an average of 71 ± 4 $m^3 d^{-1}$ from a catchment of 553 population equivalent to the 0.30 ha wetland. By comparison the Glaslough ICW is an order of magnitude larger with a functional area of 3.25 ha which treated sewage effluent for a catchment of 800 population equivalent and had a mean inflow of around 103.8 m^3 d^{-1} (Dong et al. 2011). The ratio of influent discharge (m^3 d^{-1}): functional area of wetland (ha) of the Frogshall ICW was 237: 1, and that of the Glaslough ICW was 32: 1. Similarly, an ICW installed to treat farm yard runoff in the Annestown catchment, Northern Ireland had an influent discharge (m^3 d^{-1}): functional area of wetland (ha) ratio of 8: 1 and achieved a 92% orthophosphate mean removal efficiency (Mustafa et al. 2009). Comparatively, the Frogshall ICW would have a relatively high Hydraulic Loading Rate (HLR) and lower Hydraulic Residence Time (HRT), and these factors would likely reduce nutrient removal efficiencies (Dong et al. 2011). Moreover, the Frogshall ICW was receiving a higher concentration of nutrients than the examples given here. We estimated that the ICW had a nutrient loading of 73 g m^{-2} yr^{-1} which is substantially higher than the P Assimilative Capacity (PAC) of 1 g m^{-2} yr^{-1} suggested for North American wetlands (Richardson and Qian 1999). Richardson and Qian (1999) demonstrated that wetlands operating above the PAC may suffer reduced ecosystem function and biodiversity in the long term.

There has been interest in the importance of alternative macrophyte communities, especially submerged as opposed to emergent, with respect to P removal (Brix 1994; Gu et al. 2001; Dierberg et al. 2002). Whilst macrophytes are capable of P assimilation in the range of 30–150 kg P ha^{-1} $year^{-1}$, this amount of removal is thought to be negligible in treatment wetlands receiving high P concentrations from wastewater, and the importance of macrophytes predominantly lies in the physical effects of macrophytes, for instance in stabilising sediment and increasing microbial surface area (Brix 1994, 1997). Submerged macrophytes promote additional P removal pathways which are less active in wetlands dominated by emergent vegetation, for instance by direct P uptake from the water column and by pH mediated co-precipitation of P with $CaCO_3$ (Dierberg et al. 2002). Mesocosm experiments with a hydraulic residence time of 1.5 days in the Florida Everglades have shown that submerged aquatic macrophyte communities can effectively remove SRP down to low concentrations with a with mean influent concentrations of 0.056 ± 0.003 mg P l^{-1} being reduced to 0.009 ± 0.003 mg P l^{-1} (Dierberg et al. 2002). Given that the Mundesley Beck is a chalk stream and should have high $CaCO_3$ concentrations, and that the majority of the P delivered to the Frogshall ICW is in the form of SRP, submerged macrophytes could play a role in enhancing P removal by precipitation. However, it has been suggested that submerged plant communities are less effective than emergent plant communities at treating water with high P loadings, for instance where influent concentrations are in excess of 1 mg P l^{-1} (Kadlec 2005). Therefore, whilst the high loading rates in the current ICW cells may mean that existing emergent vegetation is appropriate, if future cells were to be constructed to further attenuate nutrient concentrations, planting with submerged vegetation combined with monitoring could provide useful information about the effectiveness of submerged macrophytes in this geographic setting.

Spieles and Mitsch (2000) reported that invertebrate community composition in constructed wetlands was strongly affectly by dissolved oxygen. For instance, for the range of dissolved oxygen values 4.24–11.22 mg O_2 l^{-1} authors reported a positive Pearson's correlation coefficient of 0.74, and it was observed that lower dissolved oxygen values were associated with lower invertebrate diversity. In the current study, dissolved oxygen concentrations measured directly in the effluent from the STW were relatively oxygen deficient (6.4 mg O_2 l^{-1}) and would have the potential to suppress the diversity of the invertebrate community (Spieles and Mitsch 2000). However, by the outflow of the first wetland cell, concentrations had risen sufficiently to support a wider range of invertebrate taxa in the wetland and potentially in the receiving waterbody.

Downstream Water Quality

Whilst considerable in situ reductions in nutrient concentrations in ICWs have been demonstrated in the literature, the question of corresponding improvements in receiving waterbodies has not been as comprehensively addressed.

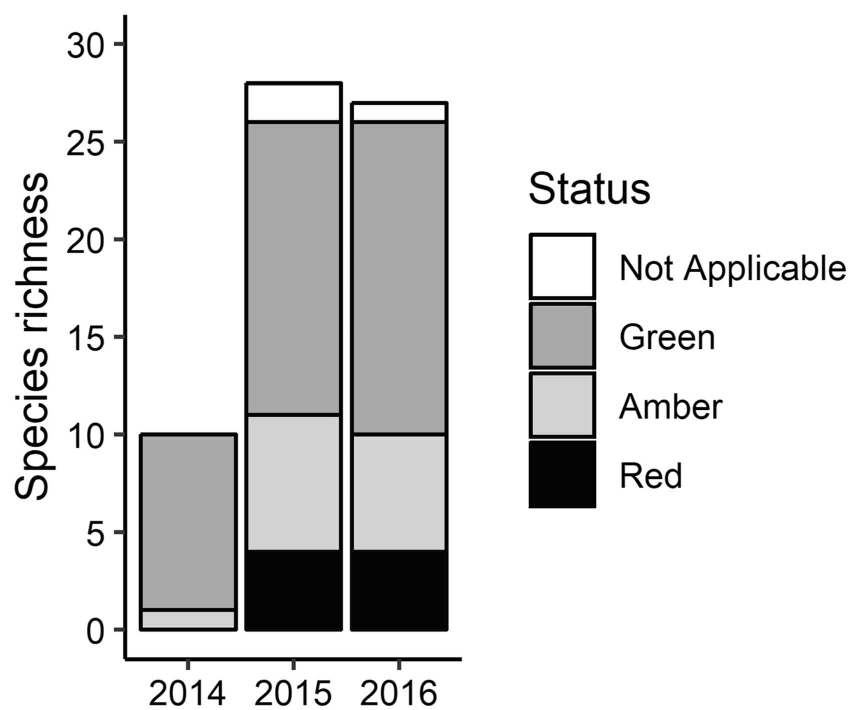

Fig. 5 Bird species, divided into BTO conservation status, recorded at the Frogshall Integrated Constructed Wetland (ICW) site, pre (2014) and post ICW construction (2015, 2016)

Mustafa et al. (2009) showed that very nutrient rich influent could be reduced to concentrations which were within recommended thresholds for the Water Framework Directive for a relatively low nutrient-status stream (receiving watercourse mean orthophosphate = 0.029 mg l^{-1} and nitrate = 0.134 mg l^{-1}). Mustafa et al. (2009) reported that the ICW discharge volume was small and a modest amount of dilution would mitigate the slightly more nutrient rich effluent ([effluent average orthophosphate = 0.94 mg l^{-1} and nitrate = 0.94 mg l^{-1}], Mustafa et al. 2009). Doody et al. (2009) monitored nutrient concentrations upstream and downstream of where an ICW discharged into a river and showed that the ICW effluent had lower nutrient concentrations than the river, and that the ICW effluent did not cause a change in downstream river nutrient concentrations, with mean river orthophosphate and nitrate concentrations reported as = 0.08 mg P l^{-1} 0.95 mg N l^{-1} both upstream and downstream.

Mean concentrations from samples taken in the 10 months prior to the commissioning of the Frogshall ICW in the Mundesley Beck had a much higher nutrient status than the above comparisons in the literature (orthophosphate = 1.23 mg l^{-1} and TON = 17.21 mg l^{-1}). After the ICW was commissioned mean TON, nitrate, TP and orthophosphate concentrations were reduced by 23–26% in the Mundesley Beck immediately downstream of the ICW. These changes were not statistically significant, but suggest that the ICW caused an improvement in the nutrient status of the receiving stream. Non-significance may have resulted from the relatively small baseline and post-intervention data sets as well as the drier conditions in the post-commissioning period which would have reduced dilution potentially causing a smaller reduction in nutrient concentrations than might otherwise have occurred if stream discharges were identical. Nevertheless, our suggestion of a downstream improvement in water quality is tentative. The increases in ammoniacal nitrogen and nitrite within the watercourse could have resulted from variations in sources to the stream, or alterations in in-stream processes, and are fairly inconsequential given the low absolute values. It is paradoxical that nutrient concentrations did not decrease more substantially in the receiving water body given that the effluent from the Northrepps STW constitutes a high proportion of the flow in the stream. It is impossible to fully understand changes in the nutrient budget, given that detailed hydrological information is not available. Data from all surface water inputs entering the stream other than the STW indicated that some sources had high concentrations of TON, higher than the ICW final effluent, but concentrations of orthophosphate were considerably lower than those found in the stream. Therefore, the small magnitude of the decrease in TON in the stream after ICW commissioning is explicable through high concentrations of TON from other sources such as the agricultural fields present on the hill slopes surrounding the stream. However, the reasons for the persistence of high concentrations of TP and orthophosphate in the stream are harder to determine. Buffering of in-stream P concentrations may have occurred due to release from “legacy” sources in stream sediment and immediate catchment (as reported for example in Sharpley et al. 2014; Jarvie et al. 2005), or due to inputs from other sources as yet not identified. The monitoring point 700 m downstream from the ICW, where the stream flows into the onstream lake, demonstrated mean reductions in TP and orthophosphate of only 9–14% and TON and nitrate of merely 5–6%. Nitrite concentrations significantly increased by 71%, although this finding may be unrelated to processes occurring in the ICW given that nitrite concentrations decreased substantially within the ICW itself. Overall, the P removal by the ICW was not sufficient to bring the stream and lake P concentrations below suggested target thresholds (Phillips et al. 2015; McCall et al. 2017). Extrapolating the relationship between the reduction in P concentrations as the ICW area increases, it was estimated that the ICW would need to be 3.6 times larger, with a total size of 1.1 ha to meet these relatively stringent standards. This estimate is tentative and based on the assumption that the relationship between ICW area and P removal will remain constant as the size of the ICW increases.

It is thought that if P concentrations are not driven below critical thresholds the algae-dominated state will persist (Scheffer et al. 1993). Nevertheless, deleterious responses to river eutrophication have been shown to exist on a gradient and therefore it is the contention of Mainstone and Parr (2002) that “any incremental reduction (in nutrient concentrations) should be seen as a positive step”. Riverine invertebrate sampling based on a Before-After-Control-Impacted experimental design has demonstrated a significantly increased abundance and biomass of riverine invertebrates downstream of the ICW after it was commissioned, although it is unclear whether the increases are related to changes in nutrient status (*van Biervliet* et al. *in prep*). More water quality and ecological data would be needed to determine whether the slight downstream reduction in nutrients is significant over the long term and whether there has been an ecological response.

Bird Species Richness

Hansson et al. (2005) suggests that the construction of wetland habitat increases biological richness through increased habitat heterogeneity. Increased bird species richness, and the appearance of BTO red list species after the construction of ICW are thus a predictable response coincident with the availability of this new habitat. Over 15,000 native aquatic macrophytes were planted within the ICW, and these colonised rapidly to reach a very high percentage coverage of two of the cells within a year of installation. Several studies found that vegetative coverage is one of the most important determinants of bird species richness in wetlands, especially in the breeding season (Sebastián-González et al. 2010; Paracuellos and Telleria 2004; Hsu et al. 2011). Aquatic macrophytes directly and indirectly provide resources for birds including food, nest construction materials and refuge from predators (Paracuellos and Telleria 2004).

Bird diversity has been shown to be positively linked with increased wetland area (Sebastián-González et al. 2010), with an optimal size exceeding 4 ha (Scott Findlay and Houlahan 1997; Hsu et al. 2011; Hansson et al. 2005; Zedler and Kercher 2005). The small size of the ICW (0.30 ha) may be a limiting factor on bird diversity. Furthermore, the very small proportion of open water, may also explain the low diversity of entirely aquatic bird species, such as Moorhen, *Gallinula chloropus*, and Kingfisher, *Alcedo atthis*, at the site. Non-aquatic bird species richness at the site also increased after ICW installation and this could be due to the other benefits provided by the wetland and surrounding habitat, such as food availability and cover. In comparison to terrestrial invertebrates, aquatic taxa are high in essential omega-3 compounds which are vital for fitness in both adult and juvenile birds and therefore the site could be offering high quality food source to birds, especially during the breeding season (Twining et al. 2016; Popova et al. 2017).

Murkin et al. (1997) found that a management regime resulting in 50:50 ratio of open water to emergent vegetation maximises biodiversity. Moreover, high nutrient concentrations within ICWs may also negatively impact on bird and invertebrate diversity (Richardson and Qian 1999; Hansson et al. 2005; Verhoeven et al. 2006). Although Becerra Jurado et al. (2010) suggest that sequential purification through ICW ponds can result in final cells capable of supporting higher levels of biodiversity.

The current study demonstrates that even small ICWs with high nutrient loadings can encourage and support a variety of bird species. Further research, including investigating over-wintering bird species, would assist in understanding the exact drivers of this increased bird diversity at the ICW, and could determine whether there could be negative consequences such as enhanced exposure to microplastics or endocrine disrupting chemicals.

Summary

Referring to the three research objectives:

(1) The ICW effluent was found to have significantly and considerably reduced nutrient concentrations and had higher dissolved oxygen concentrations relative to the influent coming from the STW.

(2) Nutrient concentrations (with the exception of ammoniacal nitrogen and nitrite) in the receiving waterbody were lower after ICW commissioning, however, this effect was not significant.

(3) Avian species richness in situ was enhanced by the ICW despite its relatively small size.

This study is consistent with findings from other authors in showing that in situ conservation and water quality benefits accrue from ICW construction (Doody et al. 2009; Harrington and McInnes 2009; Dong et al. 2011; Becerra-Jurado et al. 2014). The relatively low costs of the ICW reported here meant that the ICW was a more viable method for P removal at a small rural STW than, for instance, tertiary treatment by dosing secondarily treated effluent with metal salts to remove P by precipitation which can be expensive (Morse et al. 1998). It is probable that benefits would be more pronounced if the ICW was not sub-optimal in size and had an influent discharge (m^3 d^{-1}): functional wetland area (ha) ratio similar to examples in the literature (e.g. Harrington and McInnes 2009; Dong et al. 2011), but practical constraints precluded this.

Recommendations

There is considerable interest within the UK in the wider-scale adoption of ICWs as a method for reducing nutrient concentrations from STW effluents, with P generally the main target, but with interest also extending to the multiple benefits that such natural infrastructure interventions can provide to society. Evidence in this paper and previous studies suggest that appropriately sized and designed ICWs can have a role in this regard (Doody et al. 2009; Dong et al. 2011; McInnes et al. 2016). Whilst ICWs are cost-effective in terms of both the initial capital spend and ongoing maintenance in comparison to the alternatives for water quality improvement, it would be a mistake to consider that ongoing maintenance will not be required (Carty et al. 2008; McInnes et al. 2016). The adoption of ICWs should be undertaken in concert with careful monitoring not only to enable optimisation of design and maintenance for nutrient removal, but also to assess the dynamics of less commonly measured substances such as microplastics, pesticides and endocrine disruptors. There is already preliminary evidence, for instance, that wetlands can have an important role in substantially degrading pharmaceuticals (Li et al. 2014). If the purpose of commissioning an ICW is to improve water quality with the objective of restoring downstream ecosystems, for instance with respect to the Water Framework Directive (EC 2000), appropriate bioindicators of these ecosystems should be monitored according to a BACI, or MBACI (Muliple-Before-After-Control-Impacted) experimental design (Kibler et al. 2010; Underwood 2012). Due to the relative novelty of the ICW as a technology (the first was installed in Ireland in the mid-1990s), it is also important for regulators such as the Environment Agency in England to adopt evidence-based environmental permitting procedures for STW-ICW treatment trains which allow the potential for adaptation as the evidence base advances. Specifically with respect to the issue of permits, where the operator has control and responsibility over the entire treatment train (not the case on the Frogshall STW), we recommend: (1) the entire STW-ICW system should fall within the permitted area specified in the relevant site plan ensuring that it falls within the Management Condition which is included within Environmental Permits issued under the Environmental Permitting Regulations (UK Government 2016), meaning that the ICW will be appropriately maintained, and (2) two final effluent points should be designated within the permit, and monitored, the first as a discharge to groundwater for the effluent entering the ICW, and the second (if relevant) as a discharge to surface water where the effluent leaves the ICW.

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