

CRANFIELD UNIVERSITY

Luke Dowdeswell-Downey

Impact of the Oxford canal on water quality and ecological status of
the River Cherwell

School of Water, Energy and Environment
Advanced Water Management

MSc
Academic Year: 2020 - 2021

Supervisor: Dr Robert Grabowski
Associate Supervisor: Dr Pablo Campo Moreno
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This thesis is submitted in partial fulfilment of the requirements for
the degree of Advanced Water Management

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ABSTRACT

Good quality water provides numerous ecosystem services and it is a legal requirement of the Water Framework Directive for all waterbodies to be achieving good status. Achieving good water quality and ecological status remains a critical issue throughout most waterbodies within the UK. Waterbodies can have an impact on one another, however, determining the impact of tributaries and canal interactions on rivers remains a challenge. This study aims to investigate whether the Oxford canal has had a negative impact upon water quality and ecological status of the River Cherwell. New water quality and sediment samples were collected, upstream and downstream of where the river and canal interact at two sections at 16 sampling locations. Historical invertebrate data was examined at three locations; upstream, between and downstream of these two sections along the River Cherwell and compared to the nearby River Windrush, which does not interact with any canals. Results showed no statistical decreases in river water quality downstream of where the canal joins the river. Conversely, there was evidence that interaction with the canal water may have decreased river nutrient levels, such as nitrate, by a dilution effect. Furthermore, mixing with river water may have significantly increased orthophosphate and conductivity in the canal ($p < 0.05$). Similarly, invertebrate data showed no negative impact from the canal on the river in terms of taxa richness, BMWP, ASPT, and diversity indices. However, results suggested that the River Cherwell has had higher invertebrate metrics and indices when compared to the River Windrush. Overall, this study found no evidence to indicate that the Oxford canal negatively impacted water quality and ecological status of the River Cherwell.

Keywords: nitrate, phosphorus, invertebrate, historical data, control-impact

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Finally, I would like to thank my wife, Emily. For when I was struggling, or misunderstanding on many occasions, she was always there to make me smile, and help to explain exactly what on earth I was reading about.

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LIST OF ABBREVIATIONS

ANOVA	Analysis Of Variance
ASPT	Average Score Per Taxon
BMWP	Biological Monitoring Working Party
CRT	Canal & River Trust
DO	Dissolved Oxygen
EA	Environment Agency
LIFE	Lotic-invertebrate Index for Flow Evaluation
LOD	Limit Of Detection
NGR	National Grid Reference
PSI	Proportion of Sediment-sensitive Invertebrates
RNAG	Reasons for Not Achieving Good status
WFD	Water Framework Directive
TSS	Total Suspended Solids

Impact of the Oxford canal on the River Cherwell

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¹School of Water, Energy and Environment, Cranfield University

Good quality water provides numerous ecosystem services and it is a legal requirement of the Water Framework Directive for all waterbodies to be achieving good status. Achieving good water quality and ecological status remains a critical issue throughout most waterbodies within the UK. Waterbodies can have an impact on one another, however, determining the impact of tributaries and canal interactions on rivers remains a challenge. This study aims to investigate whether the Oxford canal has had a negative impact upon water quality and ecological status of the River Cherwell. New water quality and sediment samples were collected, upstream and downstream of where the river and canal interact at two sections at 16 sampling locations. Historical invertebrate data was examined at three locations; upstream, between and downstream of these two sections along the River Cherwell and compared to the nearby River Windrush, which does not interact with any canals. Results showed no statistical decreases in river water quality downstream of where the canal joins the river. Conversely, there was evidence that interaction with the canal water may have decreased river nutrient levels, such as nitrate, by a dilution effect. Furthermore, mixing with river water may have significantly increased orthophosphate and conductivity in the canal ($p < 0.05$). Similarly, invertebrate data showed no negative impact from the canal on the river in terms of taxa richness, BMWP, ASPT, and diversity indices. However, results suggested that the River Cherwell has had higher invertebrate metrics and indices when compared to the River Windrush. Overall, this study found no evidence to indicate that the Oxford canal negatively impacted water quality and ecological status of the River Cherwell.

Keywords: nitrate, phosphorus, invertebrate, historical data, control-impact

Project sponsored by the Canal & River Trust. Word Count: 7900

1 Introduction

Water is a natural resource of great value and as such requires good monitoring and regulation. Rivers provide numerous ecosystem services with multiple benefits for society, including provisional services, environmental regulatory services, cultural and recreational services, and supporting services, such as nutrient cycling and habitat creation (Everard, 2012a, 2012b; Medupin, 2020). The preservation and improvement of ecological status and good water quality of waterbodies within the UK, such as rivers, lakes, and canals within lowland catchments, is significant to protect these ecosystem services as well as for environmental reasons and to meet legal obligations. Identifying the drivers of poor water quality and ecological status that are contributing to the degradation of these waterbodies and therefore having a detrimental impact upon these ecosystem services is of critical importance (Chubb et al., 2014). Rivers with good water quality provide ecosystem services naturally but this may be threatened by artificial waterbodies and anthropogenic modifications (Ekka et al., 2020). Concerns are rising regarding the potential effects canals may be having on river water quality and the activities that are altering the system to exert a negative pressure on riverine ecology (Neal et al., 2006a).

The Water Framework Directive (WFD) was introduced by the European Union in 2000 for the rehabilitation of rivers and canals, in response to the widely declining ecological and chemical status of these waterbodies (Globevnik and Kaligarić, 2005). Waterbodies are classed as 'high', 'good', 'moderate', 'poor' or 'bad' based on a set of ecological and chemical quality elements (European Environment Agency, 2018). The WFD requires all waterbodies in Europe to be achieving at least good status (European Union, 2007). Rivers are required to achieve both a good ecological status and chemical status. For heavily-modified or artificial waterbodies, such as canals, the WFD acknowledges that full recovery of good water quality status may be impossible and sets the less prescriptive target of achieving good ecological and chemical potential (Everard, 2012a). Compliance with the ecological obligations of this legislation requires biological

and physico-chemical quality elements of waterbodies to be as close as possible to the undisturbed norm of the associated waterbody conditions, without anthropogenic disturbance (European Union, 2007). Over 20 years after the WFD was introduced, only 14% of UK rivers are at good ecological status and 0% for good chemical status (Gov.UK, 2021). The interaction between waterbodies, such as rivers and canals, is a key challenge in critically evaluating the reasons for water quality issues and developing management strategies to resolve these issues to improve water quality whilst maintaining anthropogenic activities.

Canals may negatively impact upon the water quality of interlinking river systems, where there are interchanges of water, particularly where the two waterbodies occupy the same channels. Generally, canals have poorer water quality than rivers, which may be due to certain aspects and activities of canals, such as slower flow conditions and intensive narrowboat traffic. Firstly, the slow flow conditions in canals leads to an almost stagnant nature where there is little reaeration, causing poor dissolved oxygen (DO) levels (Cox, 2003; European Union, 2007). DO is one of the most important factors governing the biological health of rivers and canals, such as macrophytes, fish, and invertebrates (Croijmans et al., 2021). Secondly, the use of motorised narrowboats on canals has increased recreationally, with more narrowboats in use at present than at the height of the industrial revolution (Canal & River Trust, 2020). This increase in usage increases the pressure and demand on the waterway, with increasing likelihood of oil spills, fuel leaks, and sewage leaks, contamination with heavy metals and priority substances, and increased sediment resuspension (Dodd et al., 2003). Therefore, these activities could lead to canal pollution and nutrient enrichment (e.g. nitrogen and phosphorus), and subsequently impact adjoining waterbodies.

Invertebrates are responsive to water quality variables (such as nitrate, phosphorus, DO, temperature, conductivity, total suspended solids (TSS), and pH), though invertebrate species have differing sensitivities to changes in these variables (Armon and Hänninen, 2015; Redha and Lafta Al-Zurfi, 2021). Thus, invertebrate community structure can provide a complementary indicator of

longer-term water quality conditions (Uherek and Pinto Gouveia, 2014), alongside direct water quality measurements. For example, invertebrate richness or distribution may change in response to changes in temperature and DO (Durance et al., 2009). Generally, water temperature increases during the spring and summer months, particularly in canals due to higher water residence times. Warmer waters are less able to store oxygen and so DO reduces with higher water temperature (Ficklin et al., 2013). Furthermore, higher water temperatures may stimulate algal growth (especially with nutrient enrichment), and therefore microbial respiration, resulting in a decrease in available oxygen. As such these hypoxic conditions could hamper growth rates of invertebrates as well causing lengthier development and turnover rates, resulting in an alteration of invertebrate abundance, taxa richness, and overall ecosystem processes (Ekau et al., 2010; Galic et al., 2019).

There is limited research on the effects of interactions between river and canals on water quality, so it is difficult to determine if the poorer water quality of canals has negatively impacted on the water quality and ecological status of rivers when these waters mix. Thus, the aim of this study is to investigate whether the Oxford canal is having a negative impact on the water quality and ecological status of the River Cherwell. The specific objectives are:

1. To assess physico-chemical and chemical variables from the water and sediment samples of the River Cherwell and Oxford canal, upstream and downstream of where the two waterbodies mix waters.
2. To investigate whether these river-canal interactions are impacting invertebrate biodiversity metrics and biological indices through historical data analysis, and in comparison, to a river without canal water interchange.

2 Materials and Methods

2.1 Study area

The study was conducted on the River Cherwell and Oxford canal, Oxfordshire and Northamptonshire, UK, within the Cherwell operational catchment (Figure 1) (catchment area 943 km²; (Environment Agency, 2020a)). The catchment, which is predominantly rural in character, extends from its headwaters in Charwelton, Northamptonshire and flows south towards its confluence with the River Thames in North Oxford. The collection of new water and sediment quality data and the analysis of existing invertebrate data is focused on this catchment.

The Windrush operational catchment (Figure 1) is in Gloucestershire and Oxfordshire. Like the Cherwell, the area is predominantly rural, with Witney and Carterton being the largest population mass in the catchment. Both the Cherwell and the Windrush share the Thames River Basin District. The Windrush catchment does not contain any canals, as opposed to the Cherwell catchment, and is used in this study to compare longitudinal patterns of invertebrates.

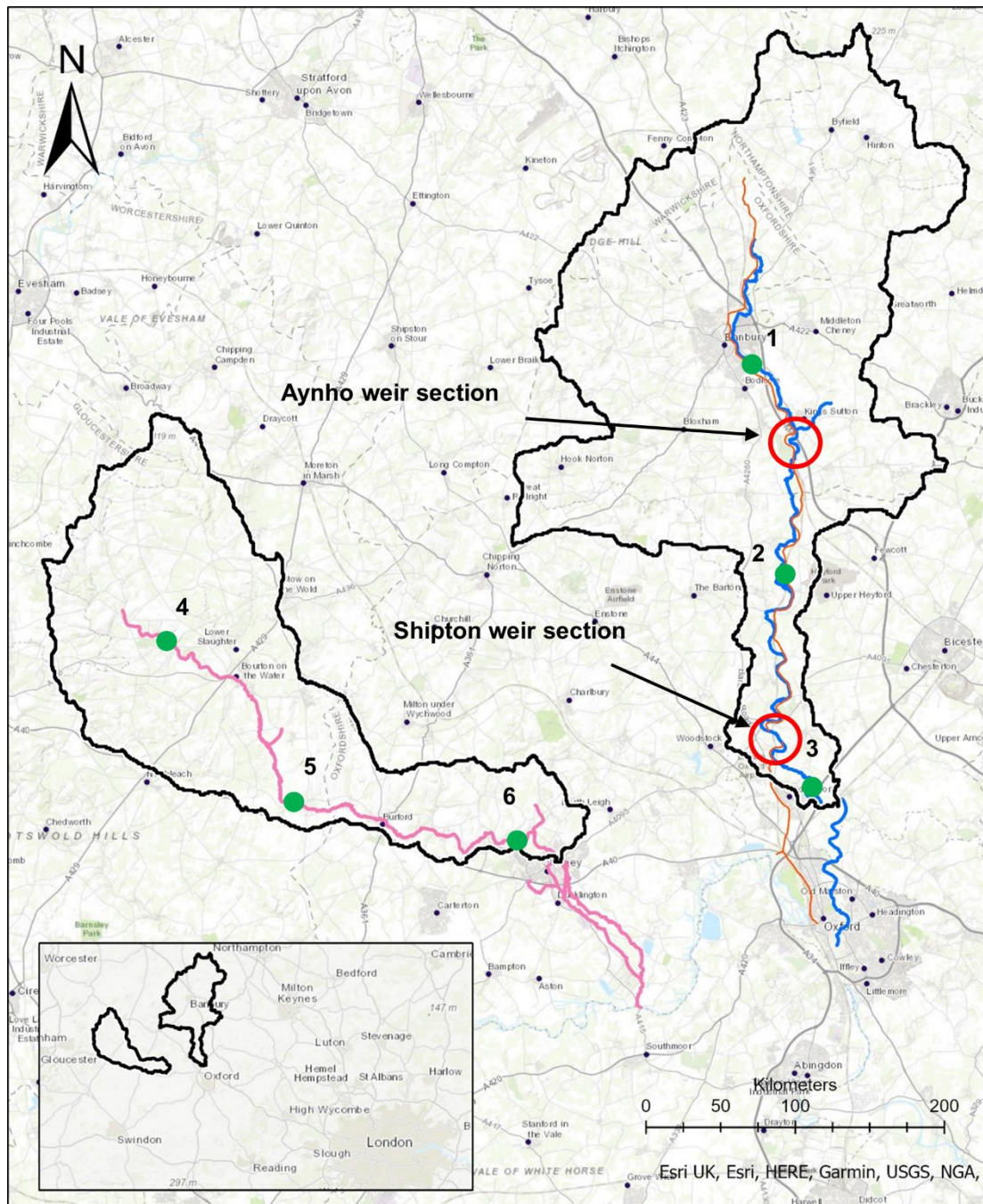
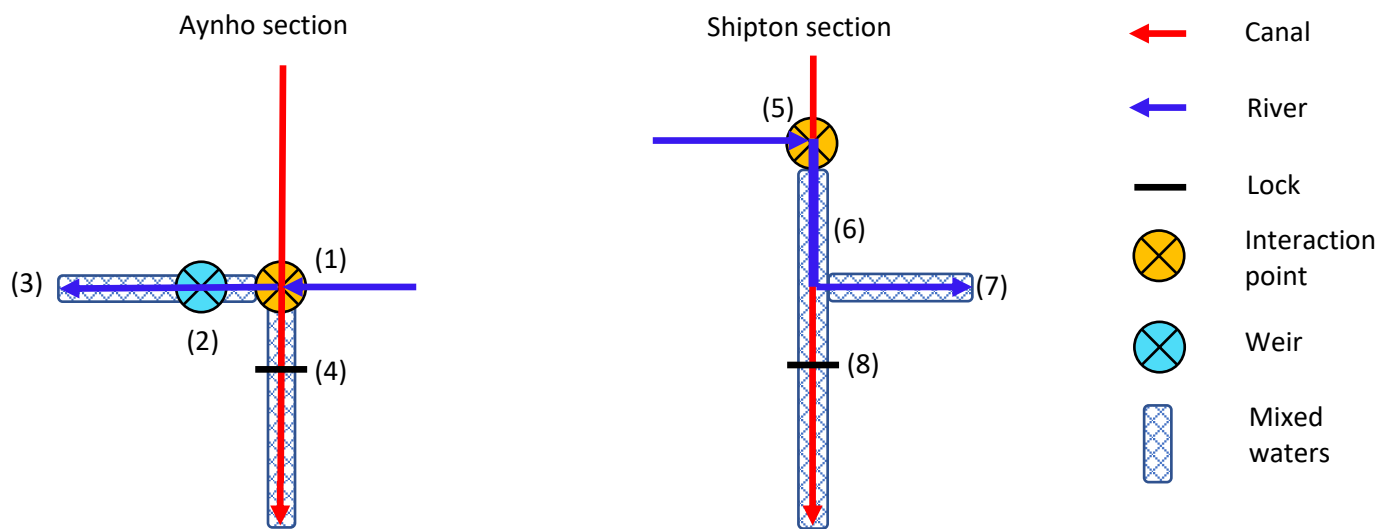


Figure 1. Map of the Cherwell operational catchment (right) with the River Cherwell in blue and the Oxford canal in red and Windrush operational catchment (left) with the River Windrush in pink. Red circles show location of interaction between the river and canal and green dots indicate Environment Agency invertebrate monitoring sites (1-6) (Environment Agency, 2020b; ESRI, 2020).

Previous records show that the majority of waterbodies within the Cherwell catchment are not meeting good status, especially due to high nitrate, phosphate and sediment levels from a range of sources such as agriculture, sewage and channel modification (Environment Agency, 2020a). In the 2019 cycle 2 of the WFD, only one out of 24 natural waterbodies met good ecological status in the Cherwell catchment but all failed the chemical status. For the same cycle for the Oxford canal, two out of the four heavily modified waterbodies achieved good ecological potential and all failed chemical potential (Environment Agency, 2020a).

The Oxford canal runs adjacent to the River Cherwell for much of its length (Environment Agency, 2020a), and directly interacts at two locations. The furthest upstream location is the Aynho section (National Grid Reference (NGR): SP 49409 33782) where the canal and river cross channels. Here the river and canal directly intersect one another, and the river exits via a weir directly opposite its point of entry (Figure 2, left). Further downstream at the Shipton section (NGR: SP 48502 17403), the canal and river occupy the same channel for a 1.5 km stretch, before the river and canal bifurcate downstream (Figure 2, right).



Ref.	Feature	Details
Aynho section		
1	River Cherwell	River flows into canal and waters mix at interaction point
2	Weir	River exits via weir opposite entry point
3	River Cherwell	River continues downstream with mixed waters
4	Aynho lock	Canal continues downstream through lock gate with mixed waters
Shipton section		
5	River Cherwell	River flows into canal and waters mix at interaction point
6	River & Canal	River and canal occupy the same channel for 1.5 km
7	River Cherwell	River continues downstream with mixed waters
8	Shipton lock	Canal continues downstream through lock gate with mixed waters

Figure 2. Schematic diagram showing the two sections of river and canal interaction with details in table explaining the flow of water. Left – Aynho section, right – Shipton section.

A control-impact survey design (Green, 1979) was used across the two sections of the River Cherwell and the Oxford canal (the Aynho section and the Shipton section, Figure 3a and b respectively). For direct water quality and sediment samples, there were 8 sampling locations in each section (4 on the River Cherwell and 4 on the Oxford canal), resulting in a total of 16 sampling locations.

Due to site access restrictions and safety, it was not possible to maintain an equidistant spacing between sampling sites. However, a minimal distance of 1 km along both the river and canal, from their confluence and bifurcation was used to ensure sample independence. Sample spacing at this distance is sufficient for studies at a meso-habitat scale to show variations in macroinvertebrates and water quality (Beisel et al., 2000). Water quality sampling took place in July 2021.

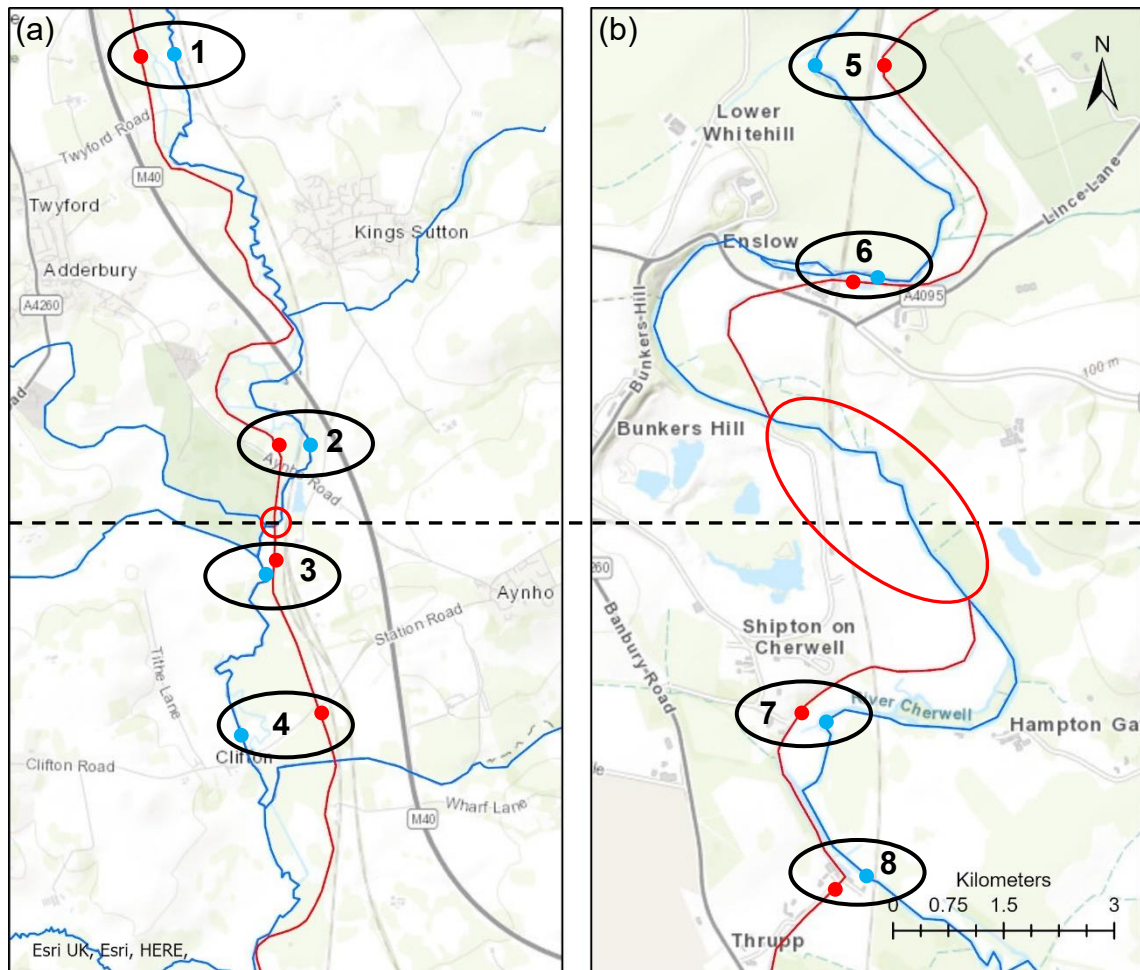


Figure 3. Maps showing the two locations of study for water and sediment, Aynho section (a) upstream and Shipton section (b) downstream, divided into two halves where the waterbodies interact, as shown by red circle and oval. River Cherwell sampling locations in blue (dots) and Oxford canal sampling locations in red (dots) from 1 to 8. (ESRI, 2020).

2.2 Water quality

Water quality variables including DO, temperature, nitrate, nitrite, phosphorus, orthophosphate, pH, conductivity, ammoniacal nitrogen and TSS, were measured in the field, or samples were collected for laboratory analysis. DO (mg/l), and temperature (°C) were obtained at each of the 16 locations (8 from the river and 8 from the canal) using a pre-calibrated, handheld multi-probe water quality meter (Hanna HI914704). A water sample was collected at each of the 16 locations in a 1l glass bottle and a 1l plastic bottle. To obtain these samples, a 3l stainless steel bucket attached to a rope was lowered into the river and canal from bridges to collect the water sample. When access from a bridge was not possible, the bucket was carefully thrown into the centre of the waterbody from the bank edge. Samples were labelled with their location and time of collection before being stored in a cool box. Samples were sent to an external laboratory (SOCOTEC, UK) and analysed through ICP-OES analysis for measurements of nitrate (mg/l), nitrite (mg/l), phosphorus as P (mg/l), orthophosphate (mg/l), pH, conductivity (µS/cm), ammoniacal N (mg/l) and TSS (mg/l).

2.3 Sediment

Sediment was investigated to assess heavy metal concentrations and priority hazardous substances, including arsenic, copper, lead, zinc, fluoranthene and mercury. For the canal, sediment samples were collected at 4 of the sampling locations with 2 samples in each section (representing upstream and downstream of the respective sections), shown as sampling locations 1, 4, 6 and 7 in

Figure 3. Collection of sediment samples was planned at 4 locations on the river, however, the river conditions (e.g. fast-flowing and deep river channel) were not suitable for this sampling method. Furthermore, alternative sites were not suitable for sediment analysis due to access restrictions, budgeting limitations, and time constraints. One sediment sample was able to be collected from the River Cherwell. Results for this sampling location can be found in 5Appendix D.

Sediment samples were collected in a 1l plastic tub, a 120 ml glass jar and a 60 ml glass jar. These samples were obtained by using a pole mounted sediment scoop or a 3L stainless steel bucket attached to rope, dependent on site access. The bucket was left to settle for two minutes on the river and canal bed. These samples were labelled with location and time before storage in a cool box. Each sample analysed at the external laboratory (SOCOTEC, UK) for measurement of arsenic, copper, lead, zinc, fluoranthene and mercury, all measured in mg/kg. 1g of air-dried and ground sample was digested for one hour with aqua regia for each sample. Once cooled the extract was filtered and pre-diluted before being analysed. Analysis was performed through optical detection by emitted light energy through ICP-OES and quantified by comparing the results against a calibration curve for each of the target analytes.

These sediment variables were chosen for analysis based on prior information taken from the Environment Agency (EA) Catchment Data Explorer (Environment Agency, 2020a) of physico-chemical, specific pollutants, priority substances and priority hazardous substances based upon the WFD cycle 2 classification quality elements.

2.4 Historical invertebrate data and analysis

Historical invertebrate data for the Cherwell was abstracted from the EA's Ecology and Fish Data Explorer (Environment Agency, 2020b) for three monitoring sites (see Figure 1): 1) EA site ID – 36232 upstream of the Aynho and Shipton sections, 2) EA site ID – 36262 between the two sections, and 3) EA site ID – 36261 downstream of the two sections. This data was used to derive a set of biodiversity metrics and biological indices for each site, as described below. These methods of water quality interpretation have previously been commonly applied in the UK to help identify water quality and ecological concerns (Hawkes, 1997; Medupin, 2020). Multiple metrics and indices were used to allow corroboration between tests, as advised by Mason (2002) and Armitage (1983).

The sites of the Cherwell were compared to assess longitudinal patterns in biodiversity metrics and biological indices, upstream and downstream of the two

sections. Longitudinal patterns in the River Windrush were also investigated, as the river has a similar catchment but does not have any river-canal mixing. The Windrush was selected for comparison because of its proximity to the Cherwell, its similar land use, and available historical invertebrate data, which enables the comparison and evaluation of whether the longitudinal patterns in the Cherwell are reflective of general patterns in the region or may be attributed to the interaction of the Cherwell with the canal. The three EA monitoring site IDs on the Windrush, running upstream to downstream are, site 4) EA site ID – 35799, site 5) EA site ID – 35507 and site 6) EA site ID – 35798 respectively (Figure 1).

Invertebrate data were obtained for the years 2013 – 2016, providing a count per family. The available data was originally collected by kick sampling, which has been widely adopted to estimate macroinvertebrate abundance due to its ease of use, cost effectiveness and speed (Sutherland, 2006). This method followed the EA's standardised method of best practice, which requires a three minute kick search, followed by a one minute hand search of larger substrate (Murray-Bligh, 2002). For the invertebrate dataset, samples were pooled across this timespan for an overall comparison, although samples had been collected during spring and autumn. Seasonal influences upon the data were tested but there were no significant differences (ANOVA, $p > 0.05$) in the biodiversity metrics and biological indices between these seasons (5E.2) and so a mean result was taken, pooling seasonal data, for each location for both rivers.

The invertebrate biodiversity metrics that were assessed include taxa richness, abundance, Inverse Simpson (Simpson, 1949), Shannon Weiner (Shannon, 1948), and evenness. Biological indices were used to interpret the dominant pressures in the rivers, including the Biological Monitoring Working Party (BMWP) (Hawkes, 1997), Average Score Per Taxon (ASPT), Lotic-invertebrate Index for Flow Evaluation (LIFE), and Proportion of Sediment-sensitive Invertebrates (PSI). The BMWP provides a score which ranks individual macroinvertebrate families on a scale of 1 to 10 based on their sensitivity/tolerance to organic pollution (i.e., nutrient enrichment that will affect DO levels) of water quality. This score is the sum of all the tolerance scores of invertebrate families within the

sample. A higher score represents better water quality (Hawkes, 1997). The ASPT was used to provide an average score by dividing BMWP by the total number of families (Armitage et al., 1983). The LIFE index was used to assess the potential impact of flow related stressors on invertebrate communities (Extence et al., 1999). A higher LIFE score is representative of faster flowing conditions. The PSI score was calculated based upon the proportion of invertebrate taxa from a sample that are sensitive to fine sediment deposition (Extence et al., 2013). This metric provides a proxy to describe to what extent British riverbeds are composed of or covered by fine sediment. Deposited fine sediment may result in an altered invertebrate community structure (Extence et al., 2013; Ryan, 1991). Lookup tables for the calculation of LIFE and PSI indices can be found in 5Appendix A.

2.5 Statistical analyses

T-tests were used to assess differences in water quality variables, comparing upstream water quality results to middle and downstream results in the River Cherwell and Oxford canal at both the Aynho section and Shipton section. Analysis of variance (ANOVA) and Tukey's post-hoc tests were used to assess if there was a statistically significant difference between biodiversity metrics and biological indices between upstream, middle, and downstream monitoring locations within the Cherwell and Windrush.

Datasets were collated in excel and analysed in R Studio (Version 1.4.1717). Results were considered significant at $p < 0.05$ (Dytham, 2011). Graphs were generated using the ggplot2 package (R Core Team, 2020). For results that were below the limit of detection (LOD), these values were substituted with a value between zero and the LOD value. For uniformity, these values were halved for inclusion within the statistical analysis (Wood et al., 2011).

3 Results

3.1 Water quality

3.1.1 Water quality of the River Cherwell at the Aynho section

Water quality results for the River Cherwell at the Aynho section did not show any significant difference between upstream and downstream locations (Figure 4, t-test, $p > 0.05$). However, there were some perceptible differences in mean nitrate concentration and DO between upstream and downstream samples. There was a reduction in mean nitrate concentration from 10.3 to 8.2 mg/l in the River Cherwell after the interaction with the Oxford canal (Figure 4a). DO dropped in the river from 10.9 mg/l upstream to 8.5 mg/l downstream, but this was not a significant decrease (Figure 4j, t-test, $p > 0.05$). There was also a very slight decrease in TSS from 8.25 to 6.25 mg/l (Figure 4h).

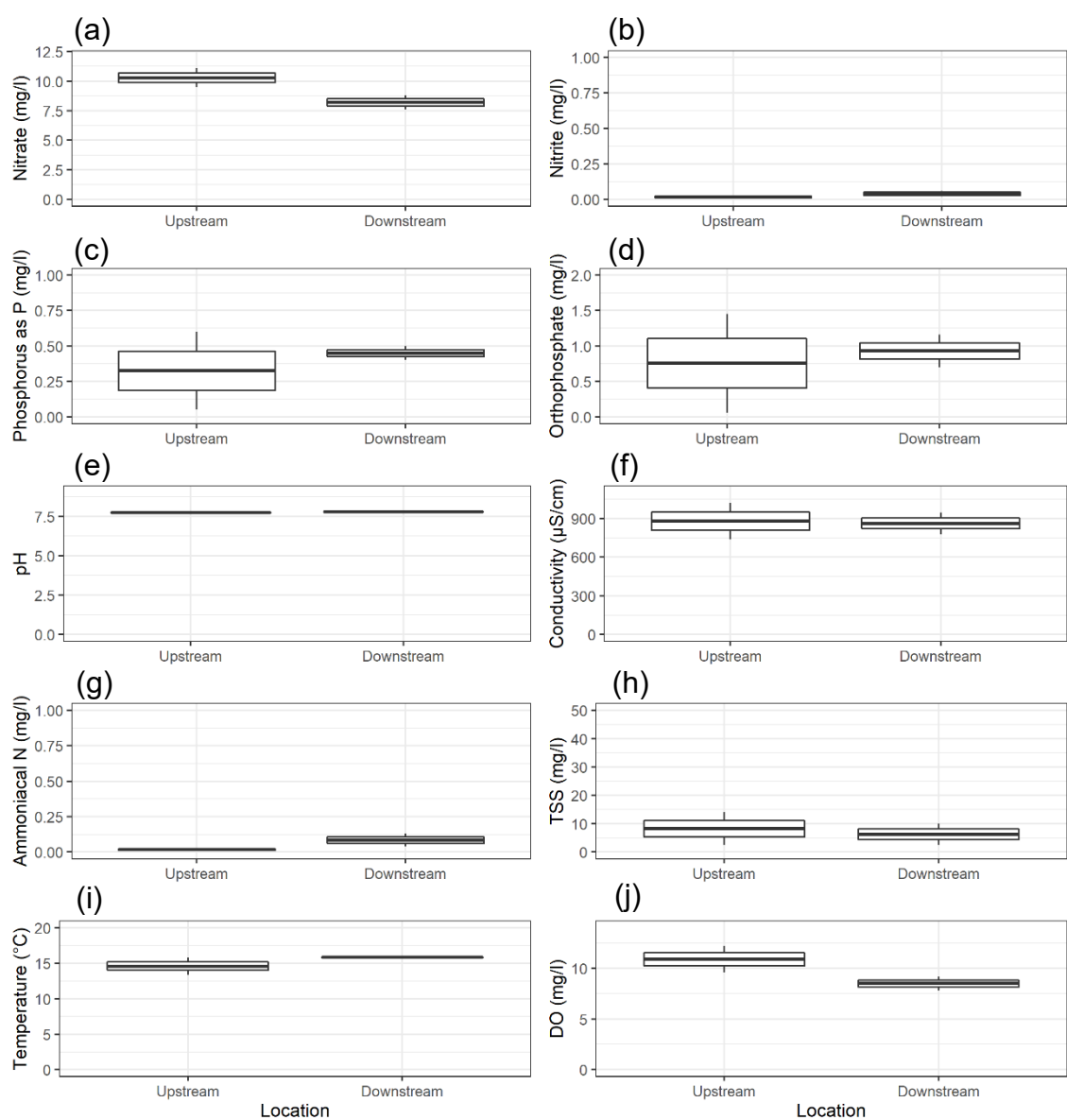


Figure 4. Physico-chemical water quality variables at the River Cherwell for the Aynho section for upstream and downstream locations.

3.1.2 Water quality for the Oxford canal at the Aynho section

Orthophosphate concentration and conductivity were statistically significantly higher downstream than upstream (Figure 5d and f, t-test, $p < 0.05$). Mean levels of orthophosphate and conductivity increased from 0.0375 mg/l upstream to 1.09 mg/l downstream and 623.5 $\mu\text{S/cm}$ upstream to 935.5 $\mu\text{S/cm}$ downstream respectively. Unlike the river at this section, there was a minor increase in mean DO, from 7.05 mg/l upstream to 8.1 mg/l downstream which was not significant (Figure 5j, t-test, $p > 0.05$). Mean TSS concentration decreased from 37 mg/l upstream to 7.75 mg/l downstream. There was an increase in mean nutrient concentrations of nitrate from 2.25 to 8.7 mg/l and phosphorus from 0.05 to 0.5 mg/l but neither were statistically significant (Figure 5a and c, t-test, $p > 0.05$).

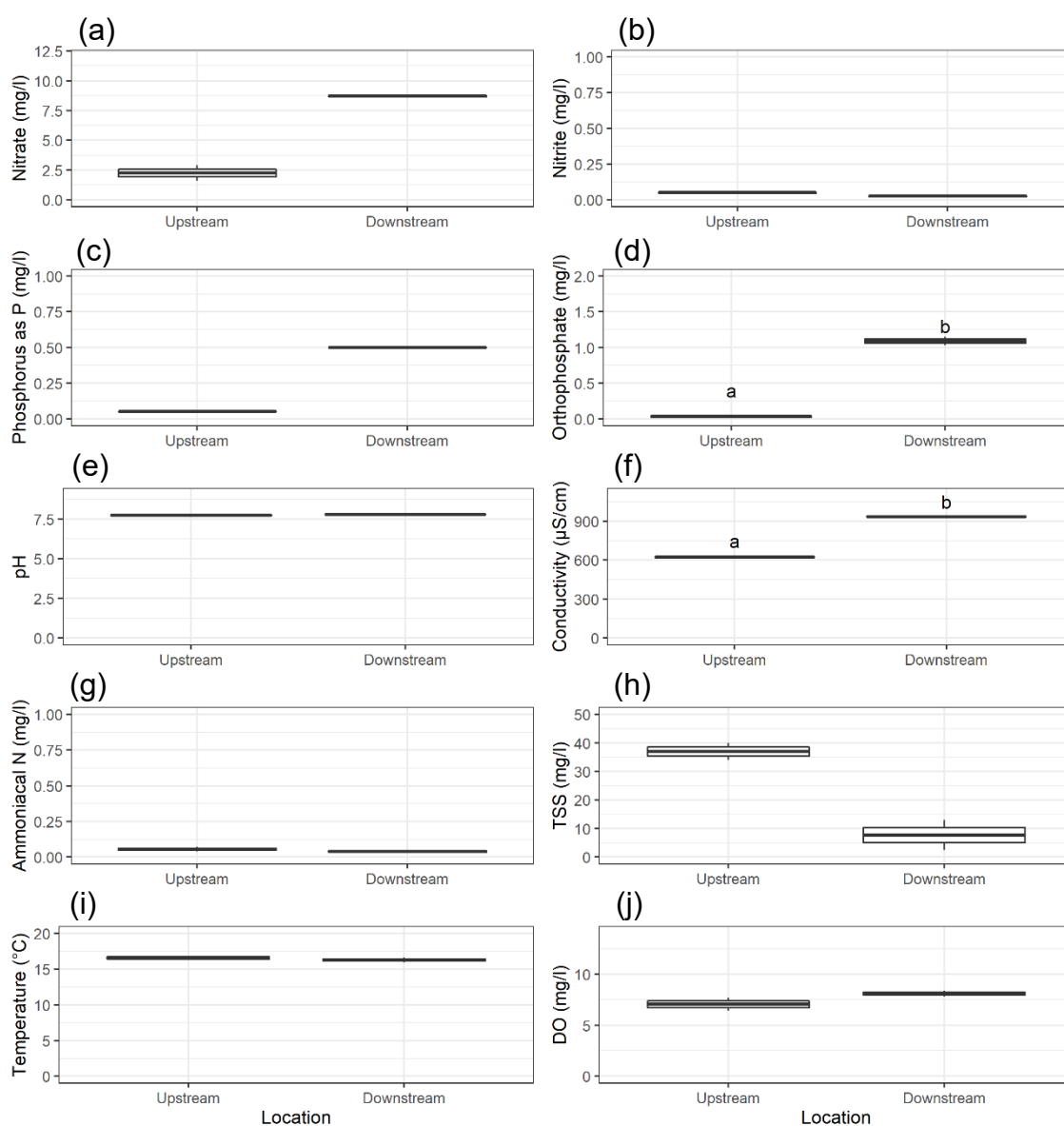


Figure 5. Physico-chemical water quality variables at the Oxford canal for the Aynho section for upstream and downstream locations. Different letters signify statistically significant difference for orthophosphate (d) (t-test, $p = 0.0193$) and conductivity (f) (t-test, $p = 00.0037$).

3.1.3 Water quality for the River Cherwell at the Shipton section

Water quality was not significantly different for any of the variables measured in the river at the Shipton section further downstream (Figure 6, t-test, $p > 0.05$). Results were very consistent across the different water quality variables before and after the 1.5 km interaction with the canal.

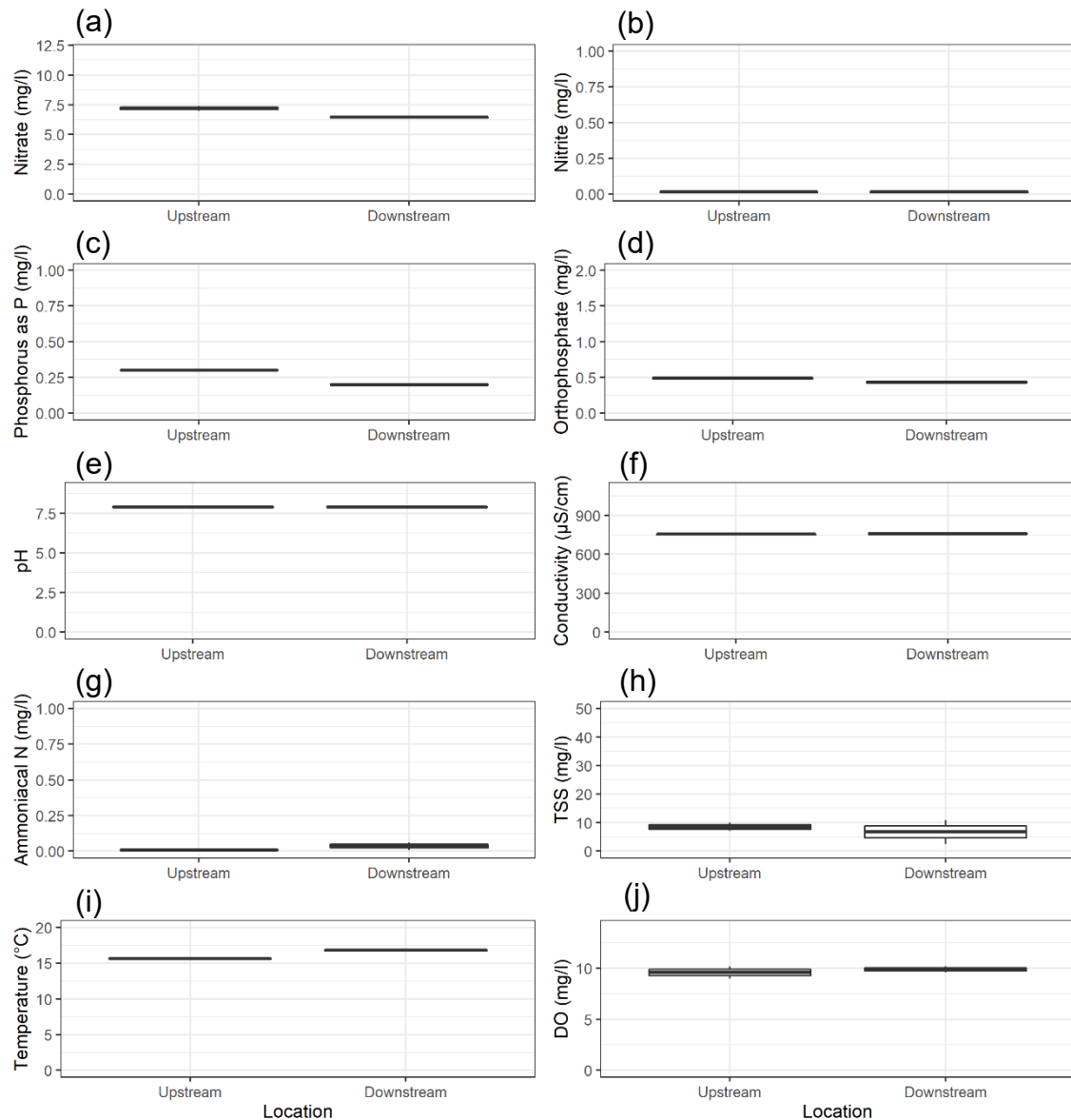


Figure 6. Physico-chemical water quality variables at the River Cherwell for the Shipton section for upstream and downstream locations.

3.1.4 Water quality for the Oxford canal at the Shipton section

There was no significant difference between upstream and downstream results at this location, except for orthophosphate, with an increase in mean result from 0.015 to 0.315 mg/l (Figure 7d, t-test, $p < 0.05$). Whilst mean TSS decreased from 20.75 to 12.25 mg/l this was not significant (Figure 7h, t-test, $p > 0.05$). Similarly, to results in section 3.1.3, results across variables were more consistent upstream and downstream with only negligible changes. However, from Figure 7 it can be seen there is an increase in the nutrients nitrate and phosphorus, although not significant (Figure 7a and c, t-test, $p > 0.05$).

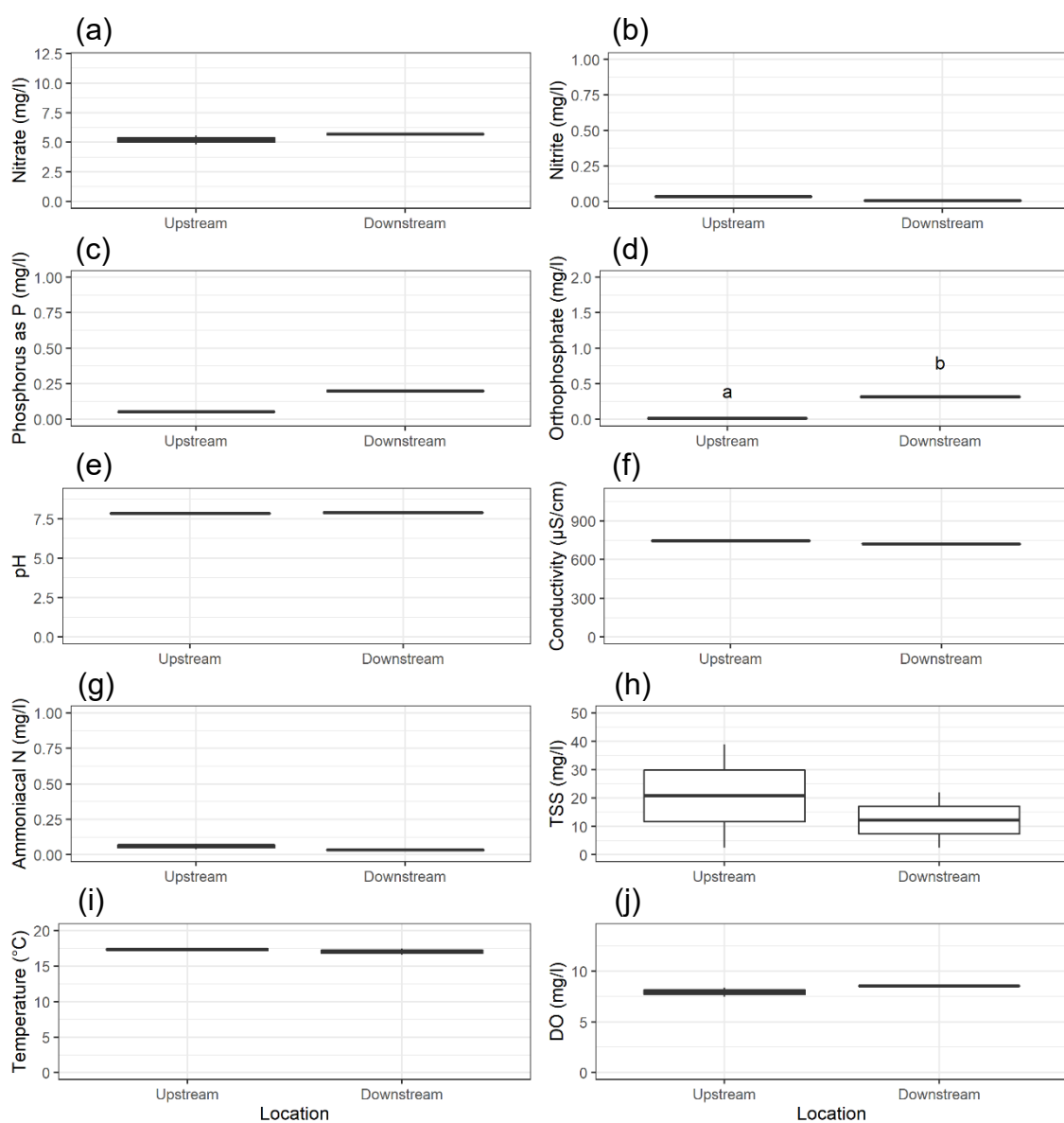


Figure 7. Physico-chemical water quality variables at the Oxford canal for the Shipton section for upstream and downstream locations. Different letters signify statistically significant difference for orthophosphate (d) (t-test, $p = 0.0106$).

3.2 Sediment

3.2.1 Sediment results of the Oxford canal at the Aynho section

Generally sediment contaminants decreased in concentration with distance downstream (Figure 8a, b, d, f), except for mercury and fluoranthene. Mercury increased from 0.025 to 3.9 mg/kg (Figure 8e) and fluoranthene increased from 0.86 to 2.52 mg/kg (Figure 8c) after the interaction with the river. Singular samples were taken so statistical significance cannot be reported.

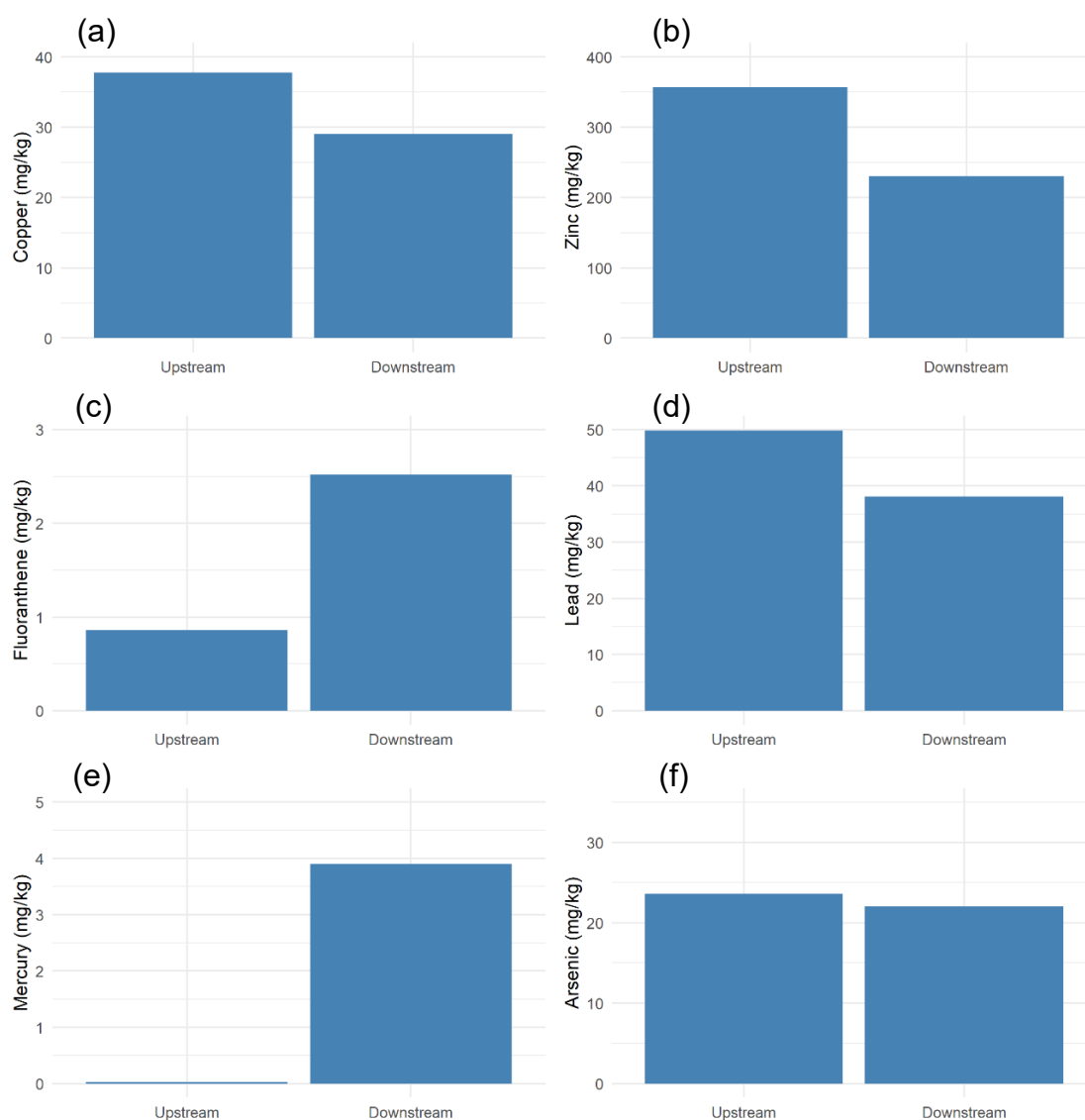


Figure 8. Concentration of sediment contaminants within the Oxford canal upstream and downstream of the Aynho section.

3.2.2 Sediment results of the Oxford canal at the Shipton section

Canal sediment contaminant concentrations were similar upstream and downstream at the Shipton section. Mercury (Figure 9e) results were 0.025 mg/kg both upstream and downstream. Statistical analysis could not be completed due to the limited number of samples.

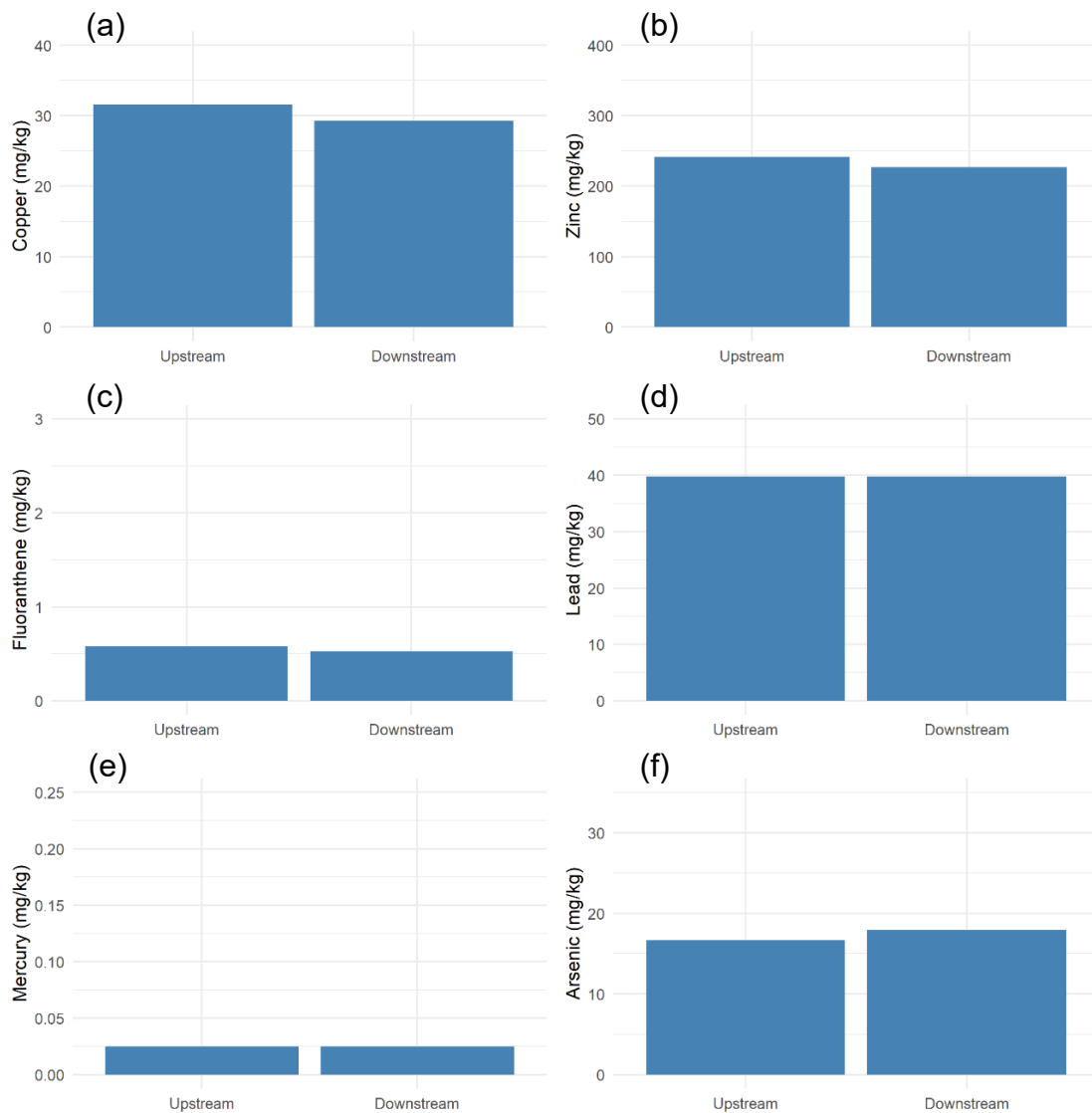


Figure 9. Concentration of sediment contaminants for the Oxford canal upstream and downstream of the Shipton section.

3.3 Historical invertebrate data

3.3.1 River Cherwell

The biodiversity metrics calculated for the River Cherwell all displayed an increase from the upstream monitoring site to the middle site after the Aynho section (Figure 10). There is some evidence of a slight decrease after the Shipton section, but this was only statistically significant for abundance (Figure 10b, ANOVA, $p < 0.05$). Abundance increased from a mean count of 242 downstream, to 1400 after the Aynho section, and then dropped to 368 after Shipton section.

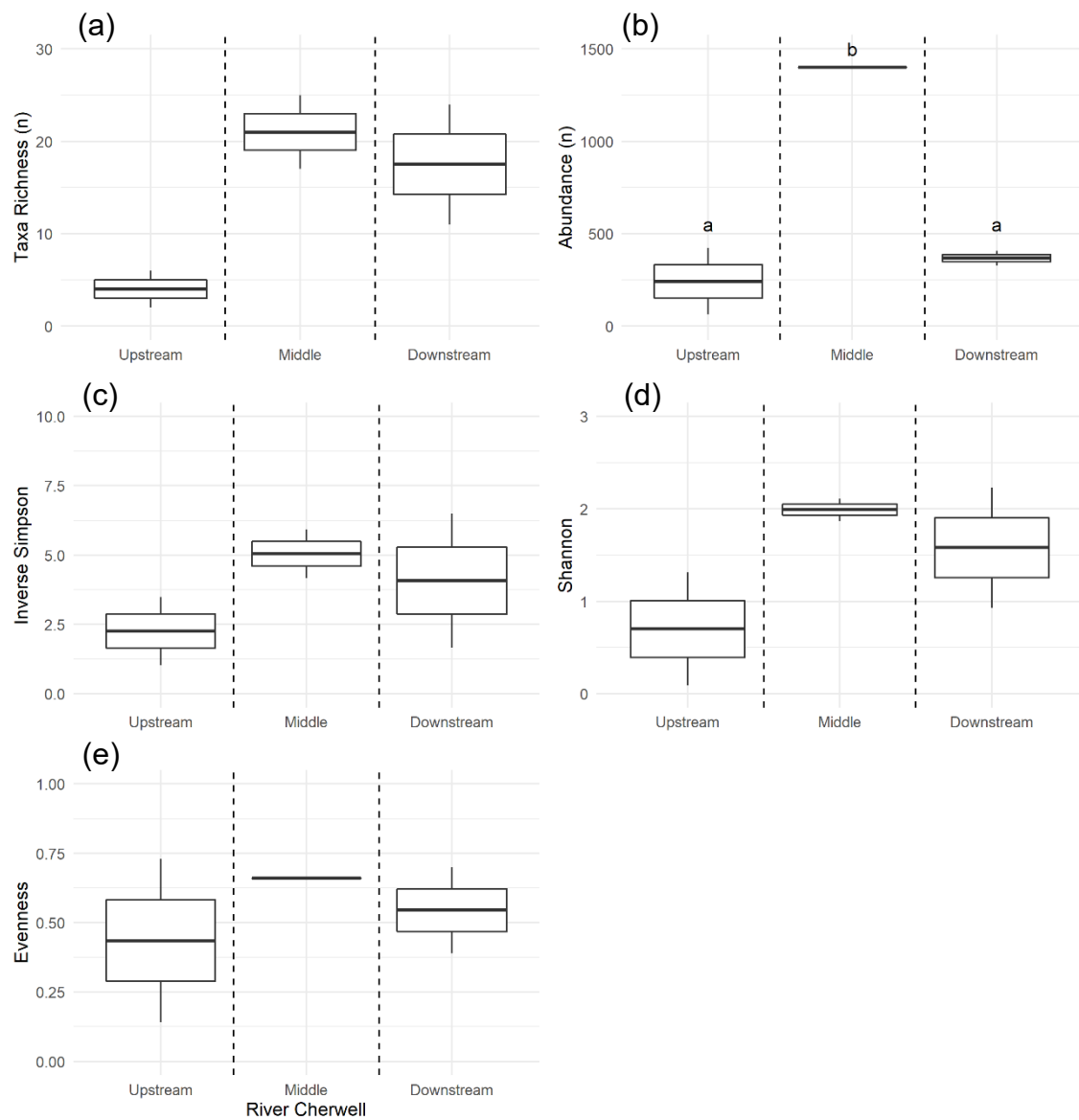


Figure 10. Biodiversity metrics of the River Cherwell, showing changes from upstream to downstream passing both the Aynho and Shipton sections (dotted lines). Different letters signify statistically significant difference for invertebrate abundance (b) (ANOVA, $p = 0.0081$).

BMWP, ASPT and LIFE were higher at the middle and downstream locations than at the upstream location. PSI scores decreased as the river flowed downstream. Every score slightly decreased after the second canal-river interaction (Figure 11), but the downstream scores remained higher than the initial upstream scores. The highest BMWP and ASPT results observed were in the middle section, between the Aynho section and the Shipton section with 116 and 6.44 respectively, although these were not different statistically (Figure 11a and b, ANOVA, $p>0.05$).

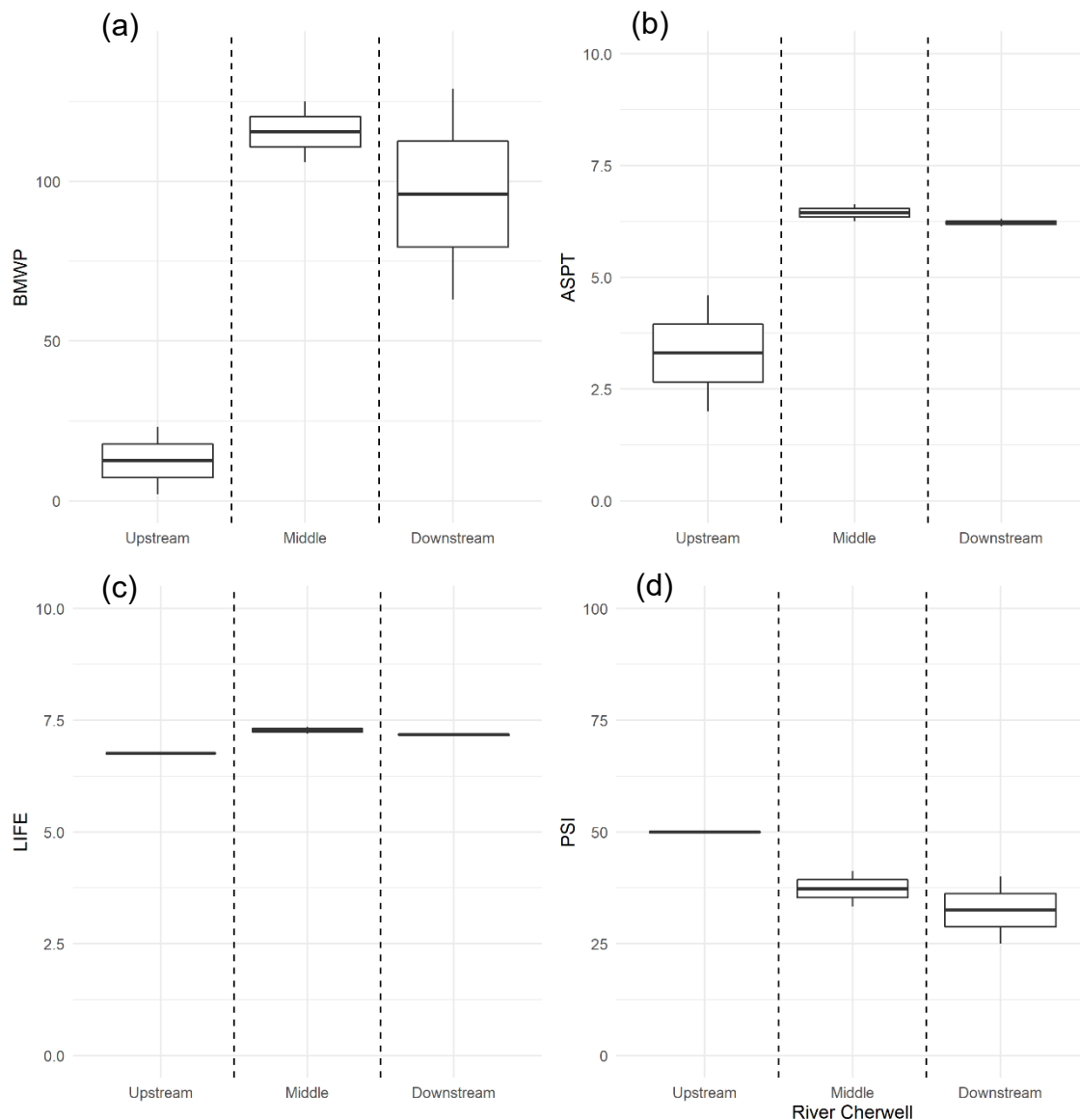


Figure 11. Biological indices of the River Cherwell. Dotted lines indicate the location of Aynho (left) and Shipton (right) sections.

3.3.2 River Windrush

For the River Windrush, the biodiversity metrics all decreased from upstream to the middle section of the river, followed by a slight increase at the downstream monitoring site (Figure 12). Abundance was statistically significantly lower downstream, (Figure 12b, ANOVA, $p < 0.05$). Abundance decreased from a mean score of 374 downstream, to 22 at the middle monitoring site and then increased

slightly to 56. Abundance at the upstream location was significantly different to the middle and downstream sections, however the middle and downstream were not statistically different from one another (Figure 12b, Tukey's test).

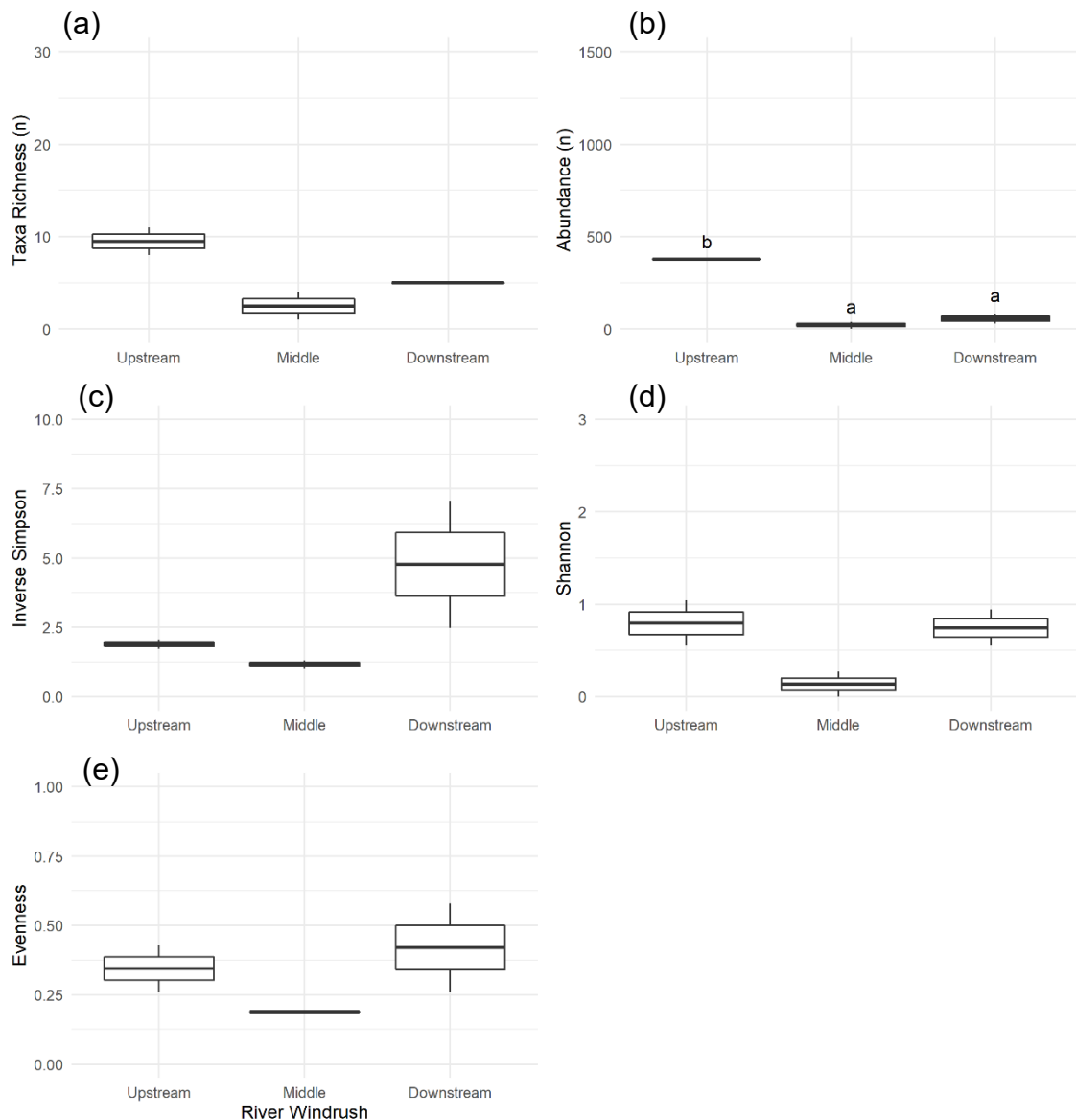


Figure 12. Biodiversity metrics of the River Windrush, showing changes from upstream to downstream. Different letters signify statistically significant difference for invertebrate abundance (b) (ANOVA, $p = 0.0015$).

BMWP and ASPT decreased from upstream to the middle monitoring site followed by an increase downstream. LIFE scores showed a similar pattern to that of the River Cherwell, however PSI scores were much higher. There was no statistical difference of for the indices within the Windrush (Figure 13, ANOVA, $p>0.05$). The highest mean BMWP and ASPT results for the Windrush were 39 (Figure 13a) and 6.66 (Figure 13b) respectively.

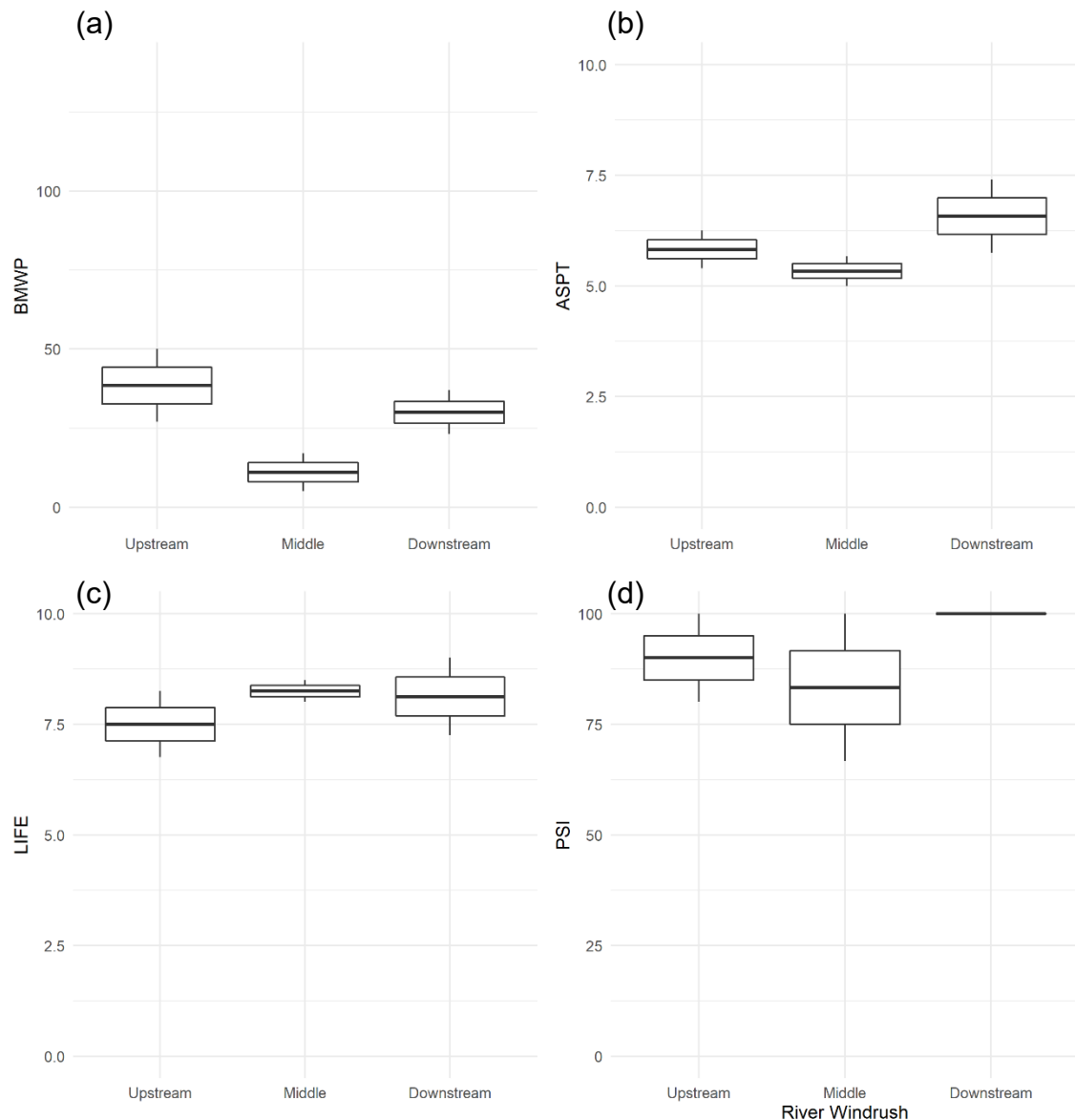


Figure 13. Biological indices of the River Windrush by location.

4 Discussion

This study aimed to evaluate whether the Oxford canal has had a negative impact on water quality and ecological status in the River Cherwell. Firstly, water quality and sediment samples were analysed in locations upstream and downstream of two sections where the river and canal interact, and secondly, invertebrate datasets were assessed in locations upstream, between, and downstream of these sections. Water quality and sediment data did not show evidence of a negative impact of the canal on the river. Conversely, the data suggested that the canal may have had a slightly beneficial impact on water quality in river. This is also supported by the invertebrate data which showed higher biodiversity metrics and biological indices in the middle and downstream sections, compared to upstream.

4.1 Water quality

There is a broad public assumption (and perhaps misconception) that canals have a generally lesser quality of water compared to rivers (Canal & River Trust, 2017). In this study, it was hypothesised that the Oxford canal has had a negative impact on the water quality of the River Cherwell. Conversely, the water quality results suggest that at the time of sampling, the Oxford canal did not have a negative effect on the water quality of the River Cherwell, and in some instances the canal positively diluted pollutants within the river. Furthermore, these results indicate that the River Cherwell may have had a negative influence upon the Oxford canal. Other studies involving water quality exchange of rivers and canals (Neal et al., 2006a) have shown a relatively uniform and consistent level of quality between the two waterbodies.

In the River Cherwell, the greatest change in nutrient concentrations occurred at the upstream Aynho section. In particular, nitrate concentration decreased following the canal-river interaction (Figure 4a) and there was also a slight increase in phosphorus concentration (Figure 4c). There were no major significant changes at the Shipton section, although concentrations of nitrate and phosphorus both decreased downstream of the section (Figure 6a and c). This

general decreasing pattern of nutrients downstream within the river may be linked to flux increase downstream as the volumetric input of water increases, creating a dilution effect with increasing flow resulting in a gradual amelioration of water quality. These downstream dilution patterns are similar to those of previous studies such as Farthing and Toetz, (2000) and Perkins and Jones, (1994), who reported declines in nitrate and phosphorus in American rivers, and Neal et al., (2005) who observed a decline in phosphorus concentrations downstream within the River Dun in the Thames basin, UK, due to increased flux. This river also has interactions with an adjacent canal and therefore may show similar trends in water quality. The relatively minor changes in concentrations at the Shipton section compared to those observed at the Aynho section could be the resultant effect from the majority of mixing occurring in the Aynho section, where the greatest increases and decreases were observed.

In the Oxford canal, the general increase in nutrient concentrations downstream, suggests that the river water had a negative impact on the canal water quality following the mixing of the waters. At the Aynho section, nitrate, phosphorus, and conductivity concentrations increased (Figure 5a, c, and f), and at both the Aynho and Shipton sections, the concentration of orthophosphate significantly increased (Figure 5d and Figure 7d). The observed increases of nutrients downstream in the canal differs slightly from the findings of other studies. The study of the River Dun by Neal et al., (2006a) observed a decrease in nutrient concentrations downstream within the Kennet and Avon canal, a canal that interacts with the River Dun. The Kennet and Avon canal did however remove (dilute) phosphorus in the river, which was found to be greater during the summer when biological activity was at its highest (Neal et al., 2006b, 2005).

The concentration of TSS in the canal showed a differing trend to the other variables, where TSS decreased within the canal following the canal-river interaction, particularly at the Aynho section (Figure 5h). Visual inspection of suspended solids on the day of sampling showed a clear difference between the canal and the river at the upper Aynho section, although there didn't appear to be significant mixing (Figure 14). Results from the canal would suggest that the river

had diluted concentrations of TSS downstream of this section (Figure 5) however, TSS concentrations in the river also decreased at this section (Figure 4). An explanation could be that as the river and canal merged, the velocity of the river decreased at this interaction, providing the opportunity for the deposition of sediment from both the river and canal. Previous studies have reported that nitrate and phosphorus concentrations increased with increasing suspended solid concentration (Jickells et al., 2000; Paudel et al., 2019; Environment Agency, 2019). However, this study suggests a differing trend, where concentrations of nitrate and phosphorus have increased, concentrations of TSS have decreased (Figure 5). These nutrients could possibly be independent of TSS if they have not adsorbed to solids and have remained in solution.

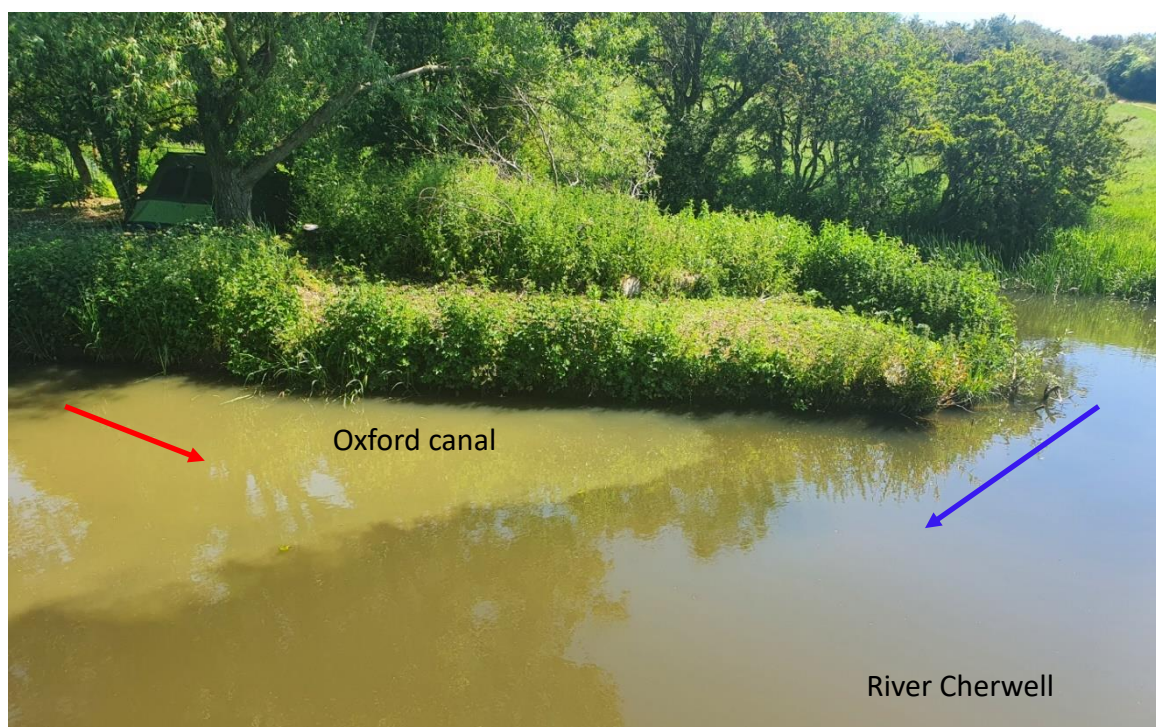


Figure 14. Photograph taken during field visit of confluence of the River Cherwell and the Oxford canal at the Aynho section showing the meeting of water and suspended solid between the two waterbodies (Source: own photo).

Whilst generally the results observed here suggest the river may have caused nutrient enrichment in the canal, an alternative explanation may be due to diffuse source pollution of nutrients or sediment. Numerous studies have stated that nutrient concentrations can be significantly impacted by diffuse sources of pollution, such as that from agriculture and rural land management (Foley et al., 2005; Valle Junior et al., 2015). Diffuse pollution is a significant issue as it could lead to potential nutrient status collapse and resulting algal blooms, as observed in the River Dun in 1998 (Environment Agency, 2000; Johnson et al., 1998; Neal et al., 2006). This pollution can be exacerbated by encroachment of livestock on waterbodies with no buffer strips (Vought et al., 1995). A photograph taken during sampling (5Appendix F, Figure F-10) shows encroachment from livestock on the River Cherwell, trampling of the riverbank and a lack of buffer strips. Bank erosion as a sediment source to rivers and canals is of major significance for water quality and riverine ecology. Sediment input to waterbodies can affect TSS and carry adsorbed nutrients, heavy metals and priority hazardous substances, affecting water quality and can create problems for invertebrates in depositional areas (Kronvang et al., 2013).

For future research a greater level of temporal resolution in sampling would be advised, as well as a more extensive use of continuous monitoring of pollution from point and diffuse sources and further work in understanding the seasonal impact of boat traffic. Concentrations of nutrients, TSS, and water quality parameters have been observed to change with time (Paudel et al., 2019) and therefore low resolution monitoring may not capture the transient nature of water quality and may miss extremes in results linked with flow and intermittent disturbance to surrounding land (Neal et al., 2006a). Additionally, a greater spatial sampling survey design would help to develop a better understanding of the locations of point source pollution within the river and/or canal. Boat traffic, which causes sediment resuspension, is highly seasonal (Dodd et al., 2003) and therefore sampling of water and sediment in the canal should take this into consideration. Therefore, higher spatial and temporal resolution may be of critical importance to develop understanding of river and canal interactions, seasonal

fluctuations in pollutant concentrations and linkages to ecological status. Furthermore, climate change and population increase are also likely to intensify the risks and impacts to river and canal water quality and ecology. As temperatures rise, flows will decrease (Neal et al., 2006a), thus leading to a reduced dilution effect of these contaminants. Agricultural runoff and erosion will intensify due to wetter winters and population increase could cause agricultural intensification of fertilisers and livestock, and therefore an enrichment of nutrients to the water.

4.2 Sediment

It was expected that the canal would have higher levels of sediment contaminants than the river because of the canal system's long history of industrial significance (Stephens et al., 2001). Sediment samples were unable to be collected in the river, and therefore the canal and river sediments cannot be directly compared. However, the canal sediment data may provide information on the catchment processes influencing heavy metal contaminants and priority hazardous substances.

Generally, sediment contaminant concentrations in the canal at the upper Aynho section decreased after the interaction between the canal and river, except for mercury and fluoranthene (Figure 8). Contrastingly at the lower Shipton section, there were no changes in sediment contaminant concentrations following the interaction between the canal and river (Figure 9). This aligns to the water quality results at the Shipton section, where there was also more stability in results between upstream and downstream locations. Canal sediments act as a sink for contaminants and as such have high concentrations of organic matter, phosphate and other substances including heavy metals and priority hazardous substances, such as arsenic, copper, lead, zinc, fluoranthene and mercury (Dodd et al., 2003). Canal sediments often show a longer-term record of locally-elevated levels of these substances which are concerning pollutants due to their high toxicity, persistence, and ability to bioaccumulate, impacting all the way through the food chain, including invertebrates (Hashim et al., 2018; Stephens et al., 2001).

Analysing sediment provides a historical record of longer-term contamination due to their greater residence times, as a proportion of these contaminants will be deposited and stored (Jarvie et al., 2005). Additionally, narrowboat traffic can cause sediment resuspension which can lead to increased exchange of these nutrients and therefore, nutrient associated management issues. Therefore, there may be a close relationship between sediment contaminant concentration and physico-chemical water quality variables. For example, phosphate exchange between canal sediment and the water column during these periods of resuspension may have a significant influence upon water quality, and phosphate is a primary nutrient that encourages eutrophication (Dodd et al., 2003).

The priority hazardous substances, fluoranthene and mercury, showed a differing trend to the other sediment contaminants at the Aynho section, where they both increased following the mixing of the canal and river waters (Figure 8c and e). These contaminants can enter waterbodies through a number of sources. For example, mercury has been used domestically and in industry, and fluoranthene primarily comes from the combustion of fossil fuels from regulated industry and road transport, which may suggest point source pollution entering the canal from nearby industry or road runoff (Environment Agency, 2019). This particular location (4) (Figure 3) had high concentrations of these contaminants. One explanation could be the increased volume of boat traffic at this location as it was a popular mooring location. This could contribute to greater combustion from powered craft and increased sediment resuspension (Dodd et al., 2003).

No statistical analysis could be done to ascertain if there was a definitive difference upstream and downstream of the two sections, due to the maximum number of samples that could be taken due to financial resources. More research on canal and river sediments needs to be undertaken to more clearly understand the association between these waterbodies. Similar to the direct water quality measurements (Section 4.1), a much higher spatial resolution of sampling would be required to further understand the potential point sources of priority substances. Furthermore, a better method of collection is also required to resolve challenges with inaccessibility to river sediment from the bank. From this study,

it would be recommended that future research makes use of sediment sampling from a boat, to access the centre of the river channel, and conduct sampling at different seasonal times of year.

4.3 Invertebrates

Invertebrates are useful indicators for the assessment of freshwater ecology and good water quality status (Moss, 1980; Xu et al., 2014), because they are ubiquitous, simple to collect, and sensitive to a broad array of environmental and chemical stressors and pollution (Armon and Hänninen, 2015). The monitoring of invertebrates, as longer-term indicators of biological quality, complement the more transient environmental (physico-chemical) variables measured from water samples (Section 4.1). As invertebrates are sensitive to these physico-chemical variables, changes in invertebrate communities may indicate an underlying longer-term trend in water quality and may provide information on the effects of canals on the water quality of adjoining rivers (Efe et al., 2012).

The biodiversity metrics and biological indices were higher in the River Cherwell than the River Windrush, suggesting that the River Cherwell had higher levels of biodiversity and a community structure more resilient to change than the River Windrush (Figures 10 to 13). For the River Cherwell, the invertebrate biodiversity metrics and biological indices are lowest upstream and show a clear increase after the interaction with the canal at the Aynho section, followed by a slight decrease after the Shipton section, though remained higher than before any interaction with the canal (Figure 10 and Figure 11). For the River Windrush, generally the biodiversity metrics and biological indices were highest upstream, then reached their lowest levels in the middle section, and then increased slightly downstream (Figure 12 and Figure 13). These longitudinal patterns suggest that the interaction with the canal had a positive impact upon ecological status within the River Cherwell.

In the River Cherwell, the PSI score was observed to decrease, which may indicate an increase in sedimentation (Figure 11d). Fine sediment accumulation generally causes degradation to ecological status and riverine habitats within

ivers. Elevated levels of sedimentation can change the suitability of the substrate for some taxa, abrade organisms and affect respiration and feeding (Harrison et al., 2007). The results for the Cherwell show the lowest levels of biodiversity metrics and indices where the PSI score is at its highest (though still moderate in the range suggesting the river was 'moderately sedimented' (Table A-4 in 5A.2). Contrastingly, in the Windrush where there were much lower overall metrics, the results showed a high PSI score in the range suggesting slight sedimentation, (Table A-4). This would suggest that sedimentation is not a primary influence on invertebrate community and abundance and may have a greater connection to wider land management strategy. Water quality results in this study indicate a reduction in TSS upstream to downstream, which does not correlate with the PSI scores of this study, however we do see a statistically significant increase in the abundance of taxa from upstream to the middle section of the river, suggesting that the Oxford canal at Aynho is not negatively impacting the ecology of the river.

The River Cherwell also displays a higher level of quality for these metrics and indices in comparison to the River Windrush and therefore a higher quality of water between the two rivers. The LIFE score suggests the canal did not have an impact on hydromorphology, however, fine sediment may have had an impact as shown by the PSI scores between the rivers.

For this study and location, there was no robust or safe way of sampling invertebrates from the riverbank, therefore historical invertebrate data was used because of the limitations of river access and time constraints. It was not possible to directly relate the observed water quality to the invertebrate results for this reason. For future studies it would be strongly recommended to perform actual invertebrate counts on the River Cherwell, at the same locations as new water quality sampling. This would allow a real time assessment of water quality in comparison to invertebrate community structure, biodiversity metrics and biological indices.

4.4 Synthesis

The overall trend of the water quality, sediment, and invertebrate results observed in this study suggest that the canal did not have a negative impact on the river. The new direct water quality samples in this study provided a real-time measurement, while the sediment and invertebrate data provided a complementary longer-term measurement of water quality and ecological status. Some studies have proposed that direct water quality measurements alone can yield sufficient accuracy and precision for many nutrient management purposes (McCarty and Haggard, 2016; Weller et al., 2010; Weller and Jordan, 2020), although, other studies have advised that these direct measurements at low resolution often fail to characterise temporal dynamics, such as seasonality or storm events (Cooper et al., 2020; Kirchner and Neal, 2013). Higher resolution water quality measurements over a longer timescale were not feasible for this study due to time and cost constraints, yet the sediment and invertebrate data provided a complementary longer-term record which showed the same trend as the direct water quality measurements. Therefore, this study shows that the canal has not had a negative impact on the river over a longer-timescale.

The EA has monitored water quality parameters, heavy metal concentrations and priority substances in sediment, and invertebrate status of the River Cherwell in 2013, 2016, and 2019. Overall, the EA classified the River Cherwell as having a moderate ecological status (Environment Agency, 2020a). Generally, the EA assessment of physico-chemical water quality parameters aligns with the patterns observed in this study, although the EA did not measure nitrate or nitrite. The EA reported decreasing phosphate concentrations resulting in a shift from poor status upstream to moderate status downstream of the Shipton section, as similarly observed here where phosphorus concentrations generally decreased with distance downstream following the mixing of waters between the river and the canal. The EA also reported consistent levels of high water quality status of pH, temperature, and ammonia, which aligns with the results observed in this study, and suggests the canal did not have a negative impact on the river. Furthermore, the EA reported an improvement in water quality status for DO, from

moderate upstream to good after the Aynho section and high downstream after the Shipton section. Whilst generally this study found no change in DO, the EA's observations support the finding here that the canal may have improved the river water quality. Data was not available for sediment in the river in this study, however the EA recorded a consistently high status for copper, iron, and zinc, and a consistently good status for fluoranthene and lead, meaning there were low levels of these heavy metals, further supporting this study's suggestion that the canal did not negatively impact the river. EA classification of invertebrates also finds a similar trend to this study, where the status of invertebrates improved downstream following the mixing of the river and canal waters, with a shift from moderate status upstream of the Aynho section to high status downstream and following the Shipton section.

Overall, the observed results here and the EA's reported water quality status, align to suggest the canal did not negatively impact the river. According to the EA, the section of Oxford canal in this study, between 2013 and 2019 was classed as achieving either a moderate or good status. In either instance, the canal is not achieving a lower level of status to the river and so supports the results of this study that the canal is not negatively impacting the canal, although it may be having a positive dilution effect. The canal has only failed its chemical status because of the recently classified priority hazardous substance, polybrominated diphenyl ethers (same failure within the river) however the canal achieved a good status for concentrations of mercury, therefore suggesting the river is not affected chemically by the canal. The negative effects on the river may instead occur from other causes, such as diffuse source pollution from agriculture and rural land management or point source pollution from the water industry. The EA identified these two categories as major sources of pollutants within the aquatic environment and are the two most significant influences of water quality and reasons for not achieving good status (RNAG) within the River Cherwell (Environment Agency, 2020a). This could connect to the livestock encroachment on the river, as well as poor nutrient management. Nutrients such as nitrate and phosphorus, as well as sediment, from sources such as agricultural fertilisers,

poor agricultural management and sewage effluent can increase biological activity, limit DO and cause undesirable symptoms of eutrophication. Of the 7617 RNAGs for phosphorus in the UK, agriculture and rural land management is the primary cause, with 3756 cases overall (March 2019) (Environment Agency, 2019). Phosphorus was not measured by the EA in the canal and so it is not possible to make direct comparisons with the results of this study. However, the results of this study assume a better quality within the canal, as phosphorus concentrations increase after interaction with the river.

5 Conclusions

In this study, no evidence was found to suggest that the Oxford canal has had a negative impact upon water quality and ecological status of the River Cherwell. Overall, the direct water quality measurements and the complementary longer-term sediment and invertebrate records of water quality supported this same pattern. These findings also corroborate those of the EA's reported water quality status for the river and canal.

Generally, the direct water quality measurements in the River Cherwell were higher downstream. These results suggest that the canal has had a positive effect on the river's water quality, possibly through a dilution effect as observed in other studies. The canal water quality generally was lower downstream, which suggests that the river had a negative effect on the canal through the process of nutrient enrichment, with the exception of TSS. Wider catchment management should be considered as a possible source of diffuse pollution through agricultural management of nutrients and livestock, which may affect nutrient inputs, erosion, and sediment.

This study was unable to directly compare the river and canal sediments. However, in the canal sediment contaminant concentrations were lower after the first interaction with the river at the Aynho section, with the exception of fluoranthene and mercury. This may be related to variations in narrowboat traffic (e.g. mooring site locations) affecting the input of heavy metals and priority

hazardous substances in the canal. Identifying the effects of canal and river mixing on sediment contaminants is a key research gap and is critical for understanding the processes affecting these contaminants in the environment and therefore the management of priority substances.

Ecological status, based on historical invertebrate data also implied that the canal has not had a negative effect on the river, but conversely may have benefited the quality of the river. Biodiversity metrics and biological indices were found to be higher downstream in the River Cherwell and were lower in the River Windrush (a river with no canal interactions), decreasing with distance downstream.

There has been a lack of research on canal and river interactions, and generally canals are expected to have had a negative impact on river water quality. This study has investigated a case study in the UK and converse to expectations suggests that the canal has not had a negative effect on the river. Further research should consider a more extensive monitoring programme to comprehensively understand the intricate relationship between these two types of waterbodies and effects of wider catchment management.

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APPENDICES

Appendix A Biotic indices lookup tables

A.1 LIFE

Table A-1. Standard EA macroinvertebrate abundance categories.

Category	Estimated abundance
A	1-9
B	10-99
C	100-999
D	1000-9999
E	10000 +

Table A-2. Scores for different abundance categories of taxa with flow groups I-VI.

Flow Groups	Abundance Categories			
	A	B	C	D/E
I Rapid	9	10	11	12
II Moderate/fast	8	9	10	11
III Slow/sluggish	7	7	7	7
IV Flowing/standing	6	5	4	3
V Standing	5	4	3	2
VI Drought resistant	4	3	2	1

A.2 PSI

Table A-3 Fine Sediment Sensitivity Rating definitions and abundance weighted scores for PSI calculation.

Group	Fine Sediment Sensitivity Ratio	Log Abundance			
		1-9	10-99	100-999	1000+
A	Highly sensitive	2	3	4	5
B	Moderately sensitive	1	2	3	4
C	Moderately insensitive	1	2	3	4
D	Highly insensitive	2	3	4	5

Table A-4. Interpretation of PSI scores.

PSI	Riverbed condition
81-100	Minimally sedimented/unsedimented
61-80	Slightly sedimented
41-60	Moderately sedimented
21-40	Sedimented
0-20	Heavily sedimented

Appendix B Statistical test results for water quality

B.1 T-test results at the Aynho section

Table B-5. Comparison of two upstream results to the two downstream results for the River Cherwell at Aynho.

Variable	p value
Nitrate (mg/l)	0.1804
Nitrite (mg/l)	0.4568
Phosphorus as P (mg/l)	0.7286
Orthophosphate (mg/l)	0.845
pH	0.5
Conductivity ($\mu\text{S}/\text{cm}$)	0.9246
Ammoniacal N (mg/l)	0.3617
TSS (mg/l)	0.8021
Temperature ($^{\circ}\text{C}$)	0.4868
DO (mg/l)	0.281

Table B-6. Comparison of two upstream results to the two downstream results for the Oxford canal at Aynho. * = statistical significance ($p < 0.05$)

Variable	p value
Nitrate (mg/l)	0.05862
Nitrite (mg/l)	0.1987
Phosphorus as P (mg/l)	-
Orthophosphate (mg/l)	0.0193*
pH	0.5
Conductivity ($\mu\text{S}/\text{cm}$)	0.003707*
Ammoniacal N (mg/l)	0.5038
TSS (mg/l)	0.06186
Temperature ($^{\circ}\text{C}$)	0.5528
DO (mg/l)	0.3265

B.2 T-test results at the Shipton section

Table B-7. Comparison of two upstream results to the two downstream results for the River Cherwell at Shipton. * = statistical significance ($p < 0.05$)

Variable	p value
Nitrate (mg/l)	0.1487
Nitrite (mg/l)	1
Phosphorus as P (mg/l)	-
Orthophosphate (mg/l)	0.1051
pH	-
Conductivity ($\mu\text{S/cm}$)	0.344
Ammoniacal N (mg/l)	0.5
TSS (mg/l)	0.7537
Temperature ($^{\circ}\text{C}$)	0.05081*
DO (mg/l)	

Table B-8. Comparison of two upstream results to the two downstream results for the Oxford canal at Shipton. * = statistical significance ($p < 0.05$)

Variable	p value
Nitrate (mg/l)	0.4296
Nitrite (mg/l)	0.06929
Phosphorus as P (mg/l)	-
Orthophosphate (mg/l)	0.01061*
pH	0.5
Conductivity ($\mu\text{S/cm}$)	0.07152
Ammoniacal N (mg/l)	0.4214
TSS (mg/l)	0.735
Temperature ($^{\circ}\text{C}$)	0.6329
DO (mg/l)	0.4076

Appendix C Statistical test results for invertebrate data

C.1 ANOVA and Tukey's post hoc tests

Table C-9. Results for difference between upstream, middle, and downstream at the River Cherwell. * = statistical significance ($p < 0.05$)

Biodiversity Metrics	
Taxa Richness	0.1471
Abundance	0.008108*
Inverse Simpson	0.5469
Shannon	0.3348
Evenness	0.7349
Management Indices	
BMWP	0.0746
ASPT	0.1031
LIFE	0.06439
PSI	0.4129

Table C-10. Results for difference between upstream, middle, and downstream at the River Windrush. * = statistical significance ($p < 0.05$)

Biodiversity Metrics	
Taxa Richness	0.05908
Abundance	0.001515*
Inverse Simpson	0.2704
Shannon	0.1651
Evenness	0.6505
Management Indices	
BMWP	0.2106
ASPT	0.4136
LIFE	0.7311
PSI	0.6219

Appendix D River Cherwell sediment results from location 4

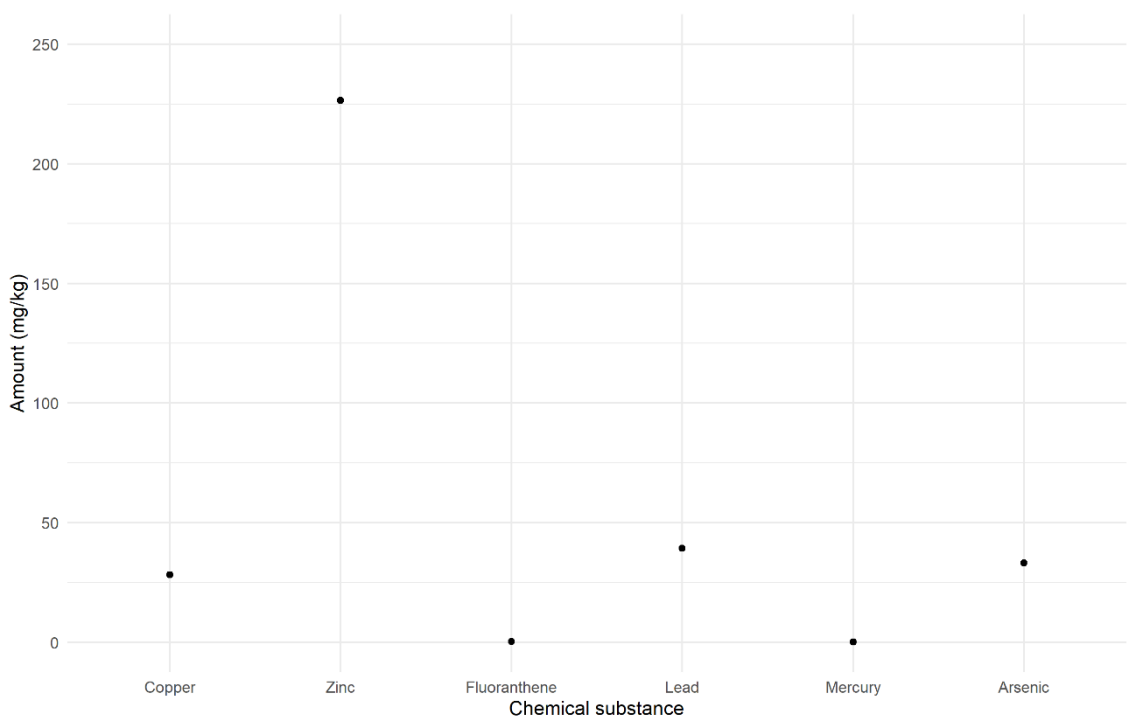


Figure D-1. Concentration of metals for the River Cherwell at sampling location 4.

Appendix E Seasonal biodiversity metrics and indices

E.1 Seasonal and locational biodiversity metrics and indices



Figure E-2. Comparison of biodiversity metrics between the River Cherwell and River Windrush for upstream, middle and downstream historical data sampling locations and for seasonal comparison. Statistical significance indicated by different letters is for location, not season.



Figure E-3. Comparison of management indices between the River Cherwell and River Windrush for upstream, middle and downstream historical data sampling locations and for seasonal comparison.

E.2 Seasonal variation between the Cherwell and Windrush

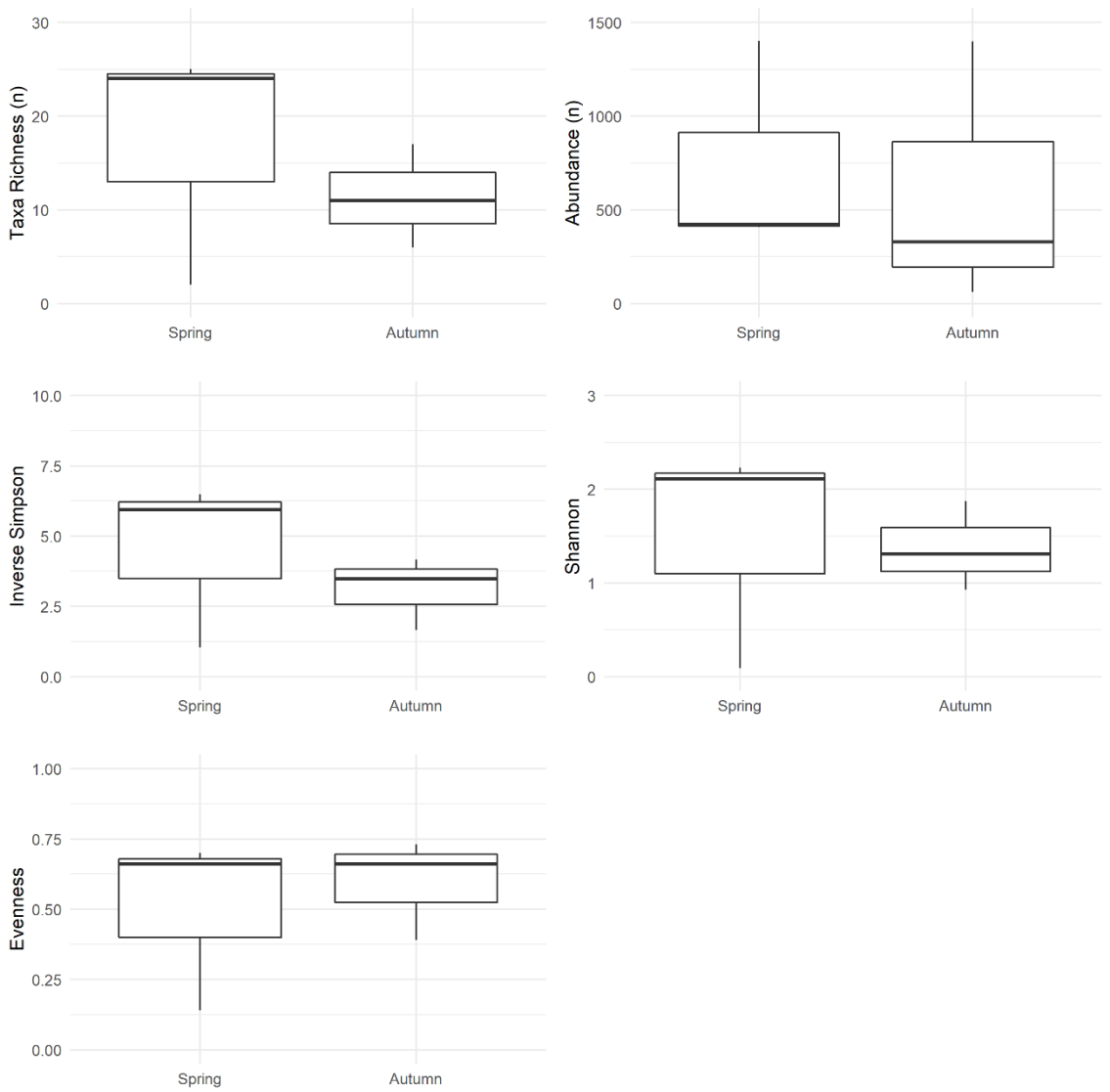


Figure E-4. Biodiversity metrics of the River Cherwell and for spring and autumn.

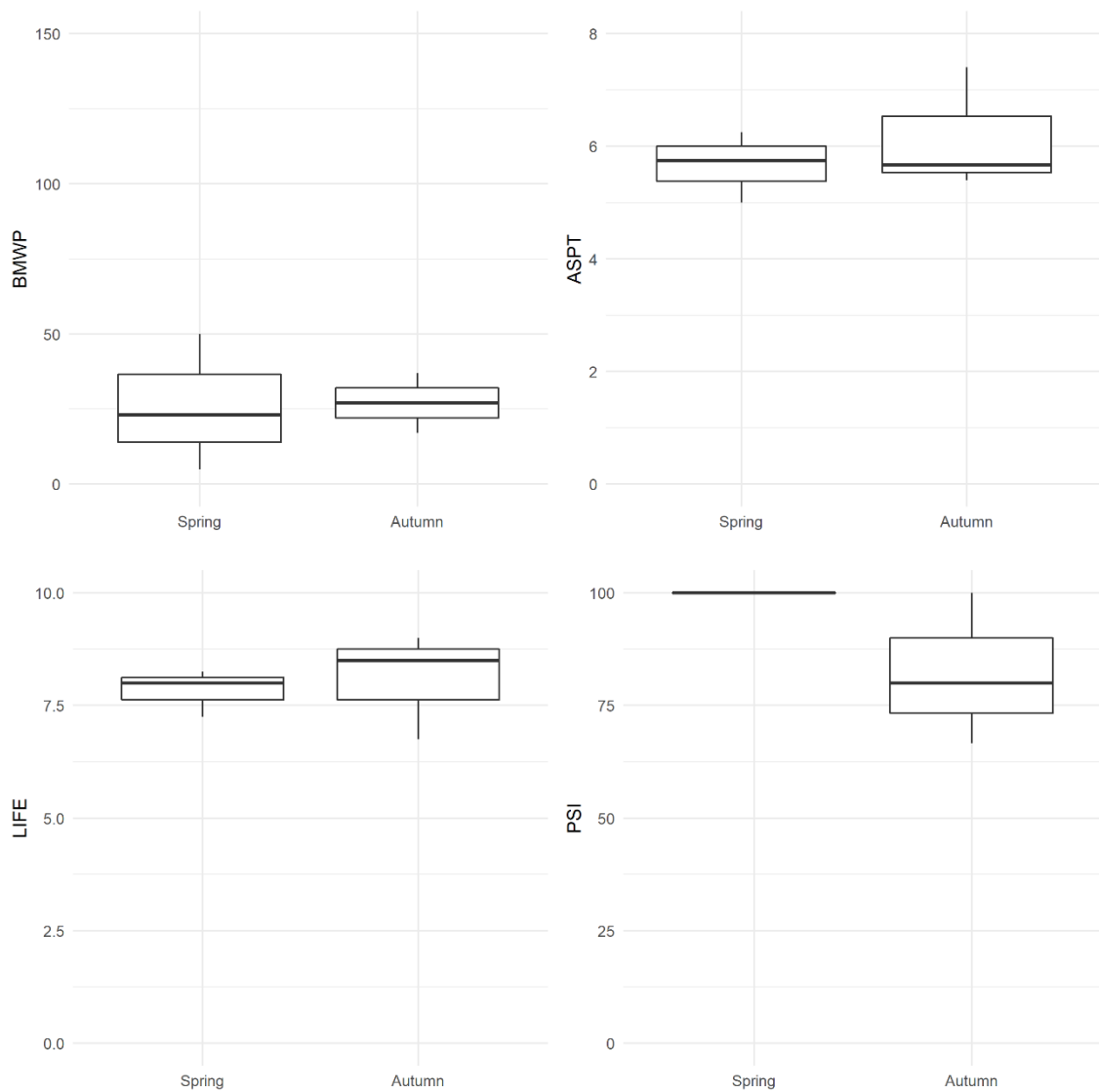


Figure E-5. Management indices of the River Cherwell for spring and autumn.

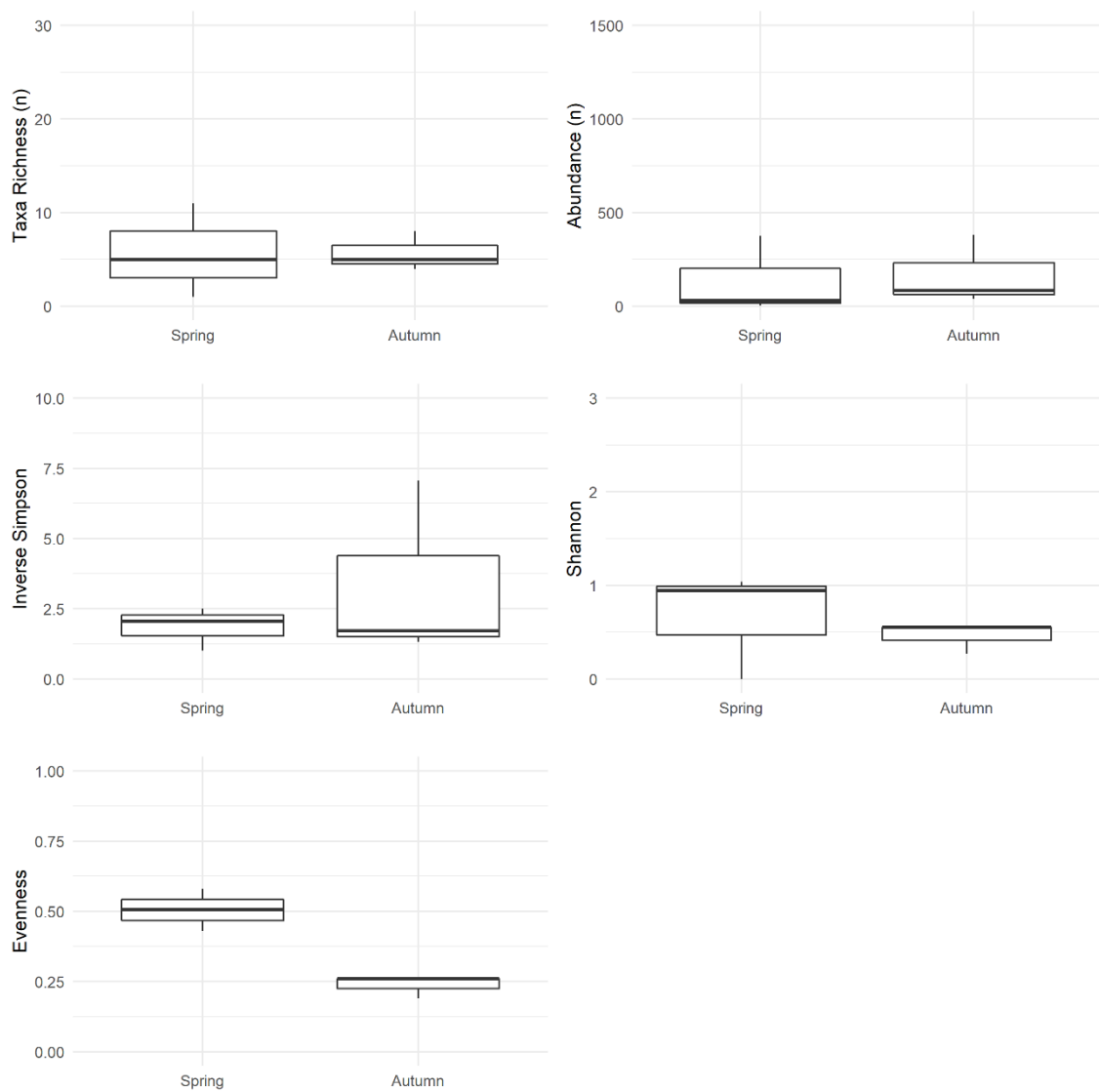


Figure E-6. Biodiversity metrics of the River Windrush for spring and autumn.

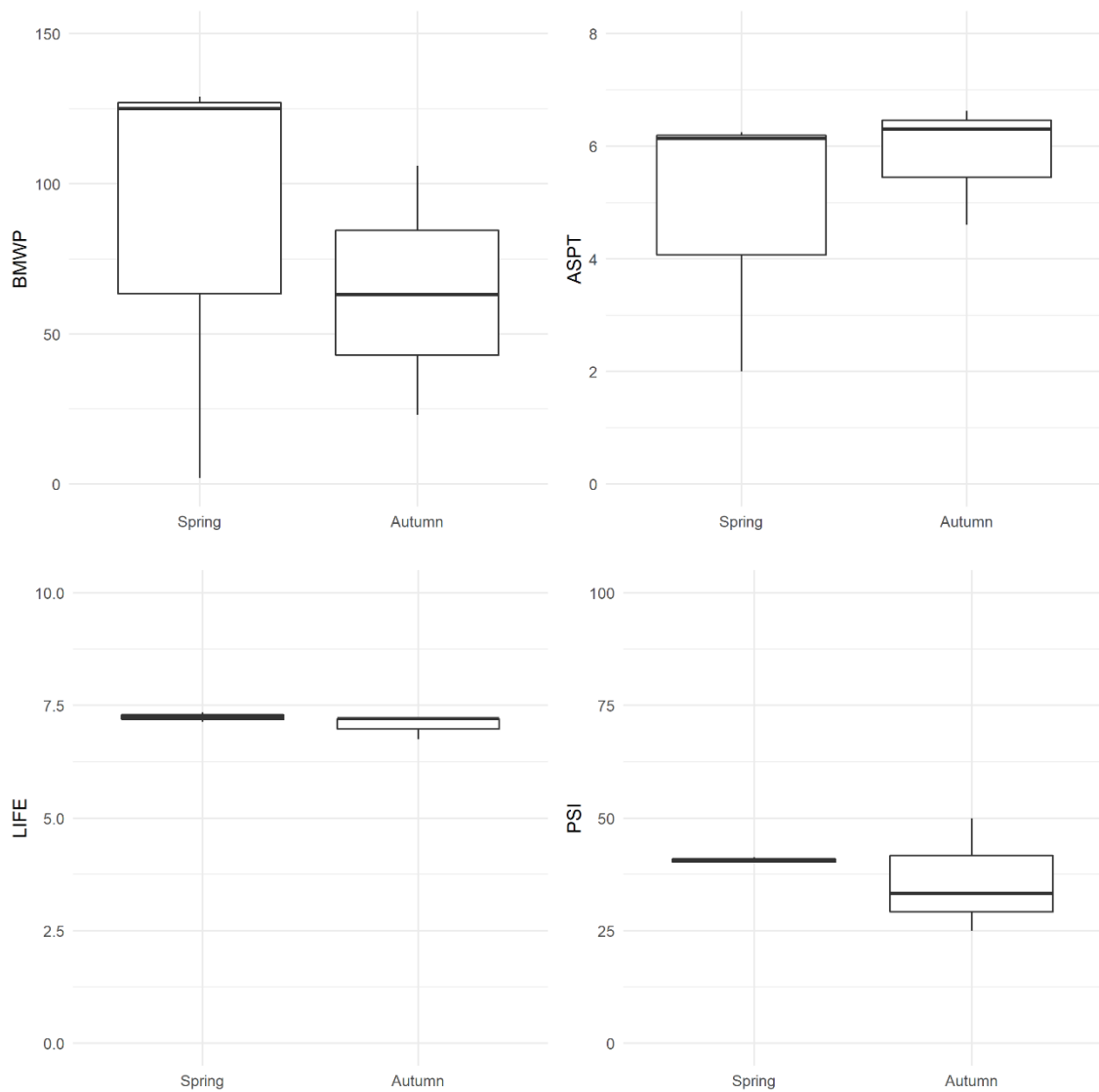


Figure E-7. Management indices of the River Windrush for spring and autumn.

E.3 Pooled data comparison of the Cherwell and Windrush

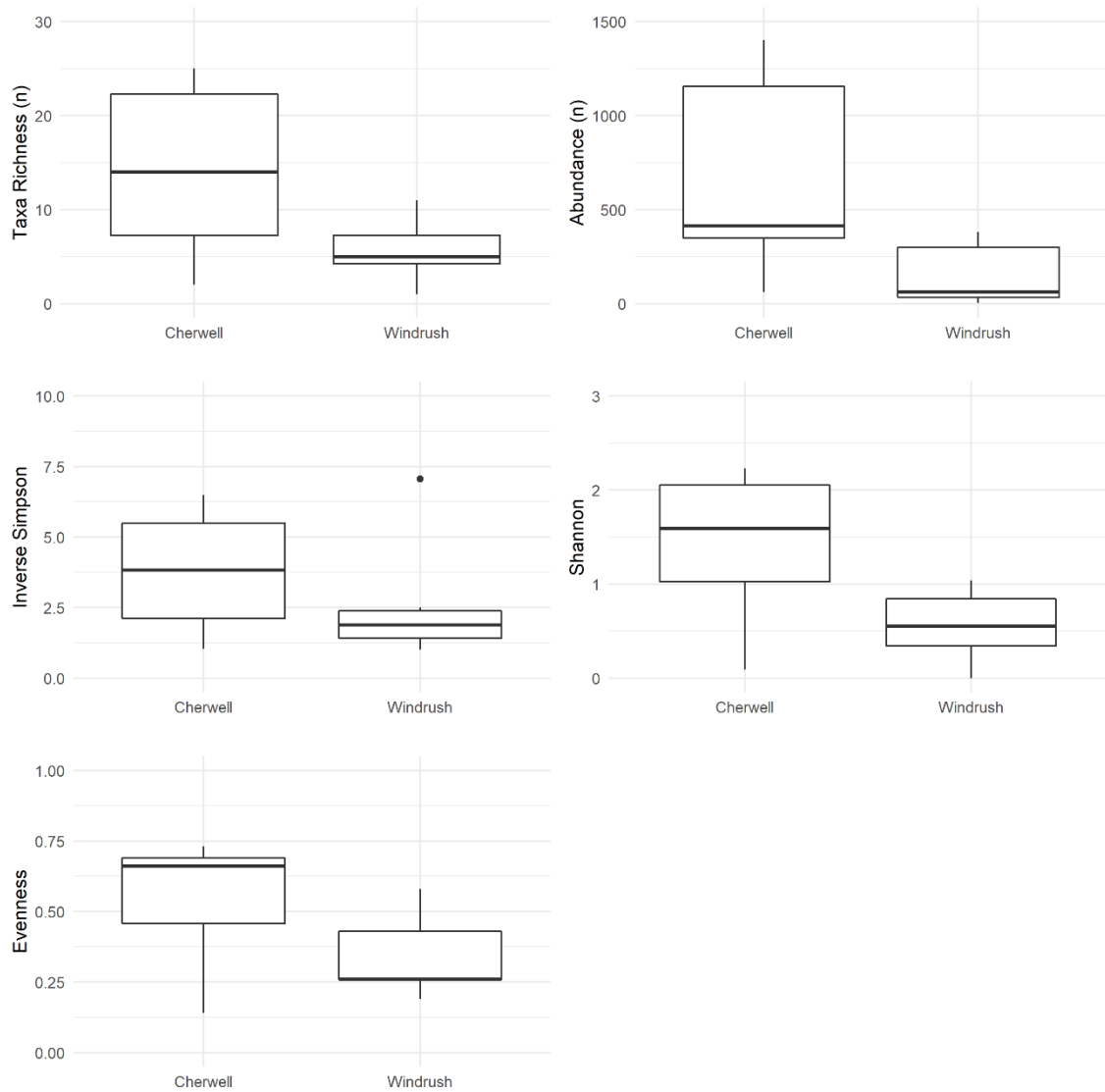


Figure E-8 The biodiversity metrics for the River Cherwell (with canal interaction) and the River Windrush (no canal interaction).

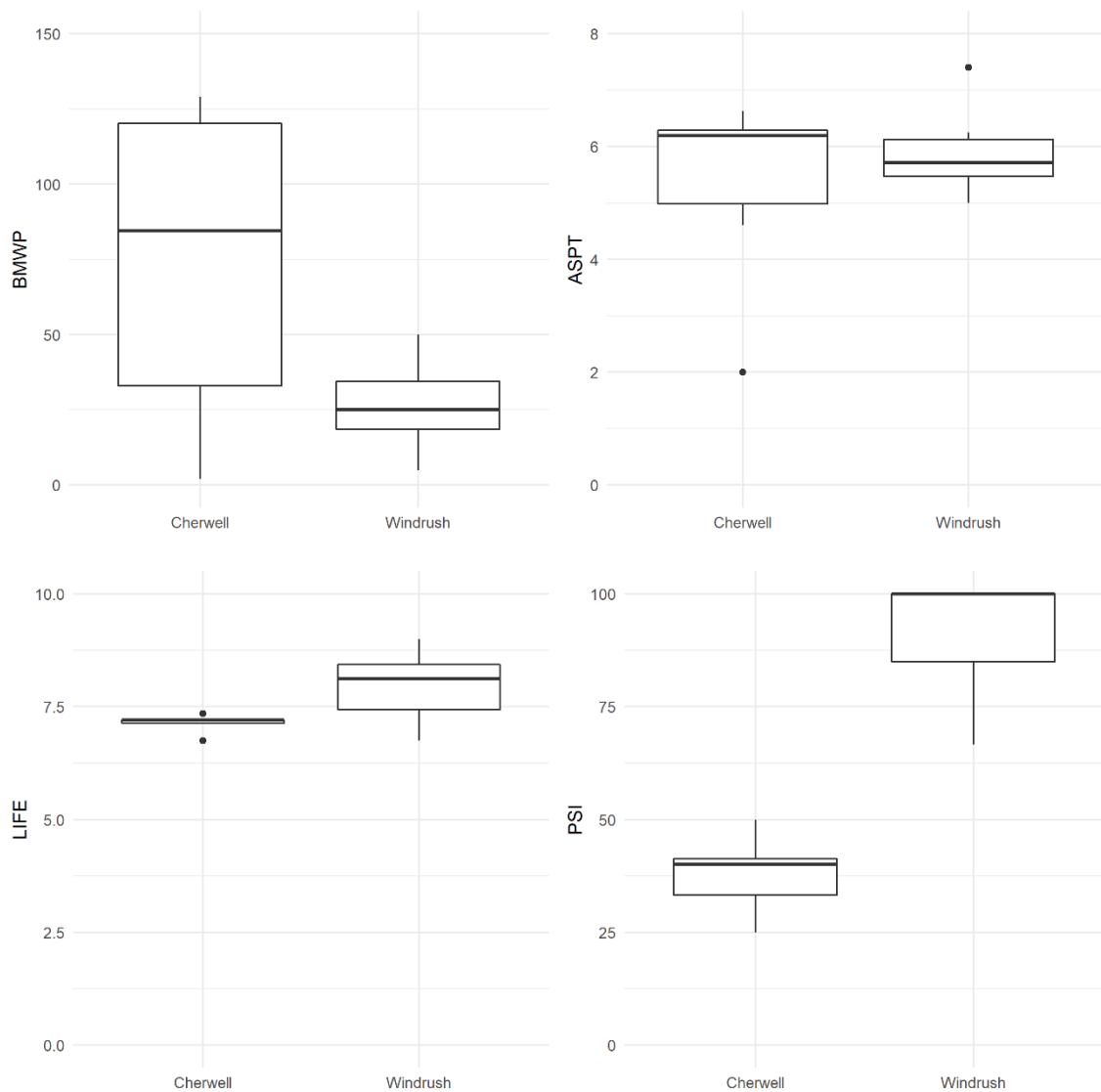


Figure E-9. The management indices for the River Cherwell (with canal interaction) and the River Windrush (no canal interaction).

Appendix F Photograph taken on day of water sampling



Figure F-10. Photograph of livestock encroachment onto the River Cherwell upstream of the Aynho section taken on the day of sampling (Source: own photo).