Developing a typology of streams in the Anthropocene: Disconnections between controls on river characteristics

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This dissertation presents the results of original research undertaken by the author. The work has been conducted in accordance with the University of Nottingham's Code of Research Conduct and Research Ethics and in accordance with the School of Geography's risk assessment procedures.

Authors signature:

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Abstract

River processes and characteristics have been substantially modified in the Anthropocene. As such, river classification systems are unsuitable for human-modified river systems because they rely on the morphological processes which may only occur in natural streams. Hence, the aim of this project was to create a new typology based on biological, chemical, hydrological and physical input categories.

Cluster analysis was used to create a typology which accounted for anthropogenic impacts and assessed the relative significance and comparability of controls on river type, using streams in the Midlands, UK to test how processes may have changed. The clusters found were compared against previous classifications and different categories of input data, then investigated to understand how anthropogenic factors may have influenced stream groupings.

Between two and three clusters were delineated for each category of variables, and a subset of river sites were found to co-occur across all variable groups. Healthy and unhealthy river types did not map on to Water Framework Directive ecological and chemical classifications, suggesting a disconnection between the Water Framework Directive and the observed morphology and condition of rivers indicated by empirical data. Categories of input data did not align with each other once cluster analysis had been performed. This inferred that system processes and responses, that the literature claimed to maintain similarities between categories, had changed.

This research has important implications for river restoration which typically focuses on one goal and category. Differences between policy- and data-driven river classifications suggest orthodox classifications may not be fit for purpose as the concept of a “natural river” does not align with how rivers are shown to respond to human influence and modification in their catchment. Therefore, research needs to focus on how rivers function in these “anthromes” and how management should respond to the disconnections between categories.

Acronym/initialisation	Full description
ASPT	Average Score Per Taxon
EA	United Kingdom Environment Agency
HMS	Habitat modification score
NRFA	National River Flow Archive
PSI	Percentage Sediment Sensitive Invertebrates
RHS	River Habitat Survey
RS	River Styles
SRS	Stream Reconnaissance Survey
STW	Sewage Treatment Works
UHS	Urban Habitat Survey
WFD	Water Framework Directive

Contents

Tables

Figures

Appendices

1. Introduction

Rivers are complex systems (Rosgen, 1994) caused by a multitude of drivers whose impact is dictated by different boundary variables creating variation (Thorne, 1997). Boundary variables then control how drivers affect river type. This is further complicated by the complete system change that has resulted in the Anthropocene (Brown et al., 2018, Brown et al., 2013, Brown et al., 2017) and the direct river management which has occurred for centuries in the UK with multiple paradigms creating complex systems of modified rivers (Downs and Gregory, 2014). Regardless, it is important to attempt to theoretically simplify the complexity of nature and the Anthropocene in order to understand the system and to manage it appropriately with the resources available (Vaughan and Ormerod, 2005).

Classification systems are needed "to usefully extrapolate system behaviour into the management arena" (Montgomery, 2001) p. 252). Classifications should be taken as conceptual, rather than direct predictions of what a river should look like, as there are too many boundary variables for a classification system to be imposed without historical analysis (Kondolf et al., 2001). There have been many seminal papers quantifying rivers (Leopold and Wolman, 1957) but these have proven more conceptual than practically useful (Thorne, 1997). More advanced classification systems such as Rosgen's have attempted to take more variables into account and describe the type of river which is present in more ways (Rosgen, 1994). In all these cases there is an element of subjectivity in the creation of boundaries between river types.

Early classification systems, as with the literature, underestimated anthropogenic impacts on the river and catchment. The Urban Habitat survey (UHS) and the modified River Styles (RS) (Brierley and Fryirs, 2013, Brierley and Fryirs, 2000) system look directly at the most urbanised rivers and how they can be typified (Davenport et al., 2004). This is an improvement, but can be seen to overly simplify the system into anthropogenic rivers and non-anthropogenic rivers. In countries such as the UK it is widely accepted that humans have influenced every catchment system (Brown et al., 2018). Therefore, it is clear there are no longer "natural" streams (Montgomery, 2008).

Deeply impacted streams are considered less functionally diverse (Ward et al., 2001) and clusters analysis was used to test this theory against the idea that anthropogenic disruption and heterogeneity can increase biodiversity and create novel ecosystems at certain scales (Acreman et al., 2014, Dufour and Piégay, 2009). No rivers were preselected to be urban or rural. Clustering algorithms allow quantitative analysis of which rivers are similar and therefore whether variables, that are dominant in controlling the typology, are morphological or anthropogenic in origin

There was a multitude of possible morphological and anthropogenic variables considered. These were split into 4 different categories to enhance the information that analysis could provide. Each category followed a strand of river science; Biological, Chemical, Hydrological and Physical metrics. These categories each formed individual cluster analyses which were comparable for final river types and the relative influence of each of these categories.

It was hypothesised that all these categories will align because of the holistic nature of river responses (Hey, 1979, Vannote et al., 1980). Traditional management assumptions further support this hypothesis, that improvements in one category will result in improvements in the others (Wohl et al., 2015). This holistic feedback mechanism might have been expected in healthy “natural” rivers however in the Anthropocene it is expected that the result may not be so clear. Furthermore, physical category has been integral to classification systems in the past however, should other categories not follow the same clustering trends (Palmer et al., 2010) then it would follow that previous classification systems have been weakened in the Anthropocene.

The knowledge created in this study will inform management based on river type. Consistently, rivers are managed for set goals (Dufour and Piégay, 2009) but rarely is this expanded to types of river. Therefore, the project will quantify the problems and opportunities UK rivers face by suggesting the river types present. Classification systems are still heavily used (SEPA, 2013) but if the morphological drivers are no longer valid in some situations then this needs to be known and accounted for accordingly.

The Water Framework Directive (WFD) dictates fresh water body management across Europe, classifying rivers on their ecological health and potential (McGinnity, 2002). In doing so it requires the Environment Agency (EA) to monitor all rivers using the same four categories, undertaken in this study, to attribute an overall assessment (McGinnity, 2002). As such the clustering from this study and the collusion between categories is directly relevant to the WFD scores. This adds considerable value to the study as the WFD has been called into question over its failure to deliver results (Hering et al., 2010). Furthermore, the methods used to measure stream quality are being questioned and by using the same data sets in this study any irregularities in data and between categories could suggest underlying issues in the drivers which contribute the WFD management.

The study will be valuable to understand the degree to which the Anthropocene has affected river type, and how this is understood by environmental managers and WFD officials.

2. Aim and Objectives

A new typology to characterise UK rivers is needed due to the range of anthropogenic influences that exert a controlling impact on river form and process. Therefore, many other river typologies and classification systems are not appropriate for streams in the Anthropocene because they rely heavily on physical descriptors of natural processes. Moreover, chemical and biological data have historically been under-utilised in river classifications, with morphological variables dominating, and often being studied independently, despite their potential significance to characterising river type (Ward and Tockner, 2001, Burt et al., 2010).

Therefore, this MRes thesis aims to develop a river typology that accounts for anthropogenic impacts and assesses the relative significance and comparability of controls on river type, including morphological, biological and chemical variables. To achieve this aim, the following objectives will be studied:

Objective 1: Do clustering methods produce clear sets of clusters for anthropogenic streams?

Objective 2: Do clusters identified in objective 1 relate to previously identified classification systems?

Objective 3: Do clusters of anthropogenic streams remain consistent when derived using different categories of input data i.e. morphological vs biological vs chemical?

Objective 4: To understand how anthropogenic factors affect site clustering

3. Literature review

3.1. Complexity in River systems

River systems are inherently complex, which makes them difficult to generalise, simplify or predict (Brierley and Fryirs, 2013). As a result, after 60 years of study since the first seminal paper on river types by (Leopold and Wolman, 1957), work still continues to attempt to partition rivers into typologies and to describe general unifying trends in river science.

Rivers are highly dynamic and complex, from scales of mm's to 100s of km's, and at timescales or seconds to millennia (Robert, 2014). For example, multiple helical flow cells define how erosion will occur on meandering river banks (Bathurst et al., 1979). There are many unknowns and unquantified controls on river form and process; for example, channel roughness is fundamental to predicting river flows but is still regularly quantified through a visual estimate of 'Manning's N' (Arcement and Schneider, 1989), which is itself a gross simplification of the variable and patchy roughness across a river bed (Attari and Hosseini, 2019).

Our fundamental understanding of river channel form can be visualised with 'Lane's Balance', which describes how an equilibrium channel is formed when the discharge and slope are 'balanced' by the sediment load and grain size (Lane, 1955). If any of the four parameters are altered, the other three will adjust towards a new balance i.e. equilibrium channel. These four driving variables are controlled by numerous boundary variables, which define how these drivers are implemented in different hydro-geomorphic contexts (Brierley and Fryirs, 2013). These boundary variables can change with season, such as vegetation (Hickin, 1984), or with precipitation levels (Dapporto et al., 2003, Taylor et al., 2000). Therefore, a river channel represents the biome within which it is situated and any change in that biome will be manifest in the dynamics of the river (Johnson et al. 2019). Anthropogenic alterations to river channels prevent rivers from responding to changes in their catchment / biome and so, they can be out of balance i.e. in disequilibrium with the catchment controls and drivers. In such scenarios, humans can become a key driver and control on river processes, adding further complexity to understanding river form and dynamics.

3.2. Human impacts on rivers

The Anthropocene refers to the era in which humans have become major geological drivers, in some cases having taken control from natural processes (Crutzen, 2006). Indeed, many river forms and processes that were considered natural, such as meandering planforms and the basis of hydraulic geometry, are now understood to be representative of human impact. For example, in America the alteration of rivers through the installation of mills, changed river type significantly (Walter and Merritts, 2008), resulting in the complete removal of "natural" streams (Montgomery, 2008). These mills are shown to leave a legacy in sediment and slope dynamics that lasts considerably longer than the mill itself (Poeppl et al., 2015). In the UK, evidence suggests no natural streams remain in the Anthropocene, attributed to an increased run off of water and sediment from farming and historic catchment deforestation (Brown et al., 2018). In these cases, there are multiple drivers due to the co-existence of land-use change (e.g. farming intensification) and direct river alteration (e.g. mill dam construction) (Brown et al., 2013, Brown et al., 2017). The rest of this section will highlight the main anthropogenic impacts on river systems and how these may alter river type.

A suite of anthropogenic drivers control hydro-geomorphic change (Goudie, 2016), which act in unison or counteract each other (Downs et al., 2013) to create a continuous scale of river types. Direct river management has changed over time (Table. 1.) with multiple different management paradigms often impacting the same streams (Downs and Gregory, 2014). Management is spatially, as well as temporally, variable with different social and economic contexts resulting in different management options (Downs and Gregory, 2014).

Table. 1: Table of the six chronological phases of river use and the managements that relate to these from (Downs and Gregory, 2014).

Chronological Phase	**Characteristic developments**	**Management methods**
Hydraulic civilizations	River flow regulation	Dam construction
	Irrigation	River diversions
	Land Reclamation	Ditch building
		Land drainage
Pre-industrial revolution	Flow regulation	Land drainage
	Drainage schemes	In-channel structures
	Fish weirs	River diversions
	Water mills	Canal construction
	Navigation	Dredging
	Timber transport	Local channelization
Industrial revolution	Industrial mills	Dam construction
	Cooling water	Canal building
	Power generation	River diversions
	Irrigation	Channelization
	Water supply	
Late 19th to mid-20th century	River flow regulation	Large dam construction

	Conjunctive and multiple use river projects Flood defence	Channelization River diversions Structural revetment River basin planning
Second part of the 20th century	River flow regulation Integrated use river projects Flood control Conservation management Re-management of rivers	Large dam construction River basin planning Channelization Structural and bioengineering revetments River diversions Mitigation, enhancement and restoration techniques
Late 20th and early 21st centuries	Conservation Re-management of rivers Sustainable use river projects	Integrated river basin planning Re-regulation of flow Mitigation, enhancement and restoration techniques Hybrid and bioengineered revetments

Rural rivers are often considered to be more natural than urban rivers but, remain heavily impacted by people, with an associated loss of habitat and hydrological heterogeneity (Negishi et al., 2002). Rural streams are often straightened to improve drainage, reduce flooding and create fields that easier to be worked (Downs and Gregory, 2014), meaning that their sinuosity and bank shape is significantly more uniform than would be found naturally (Prior, 2016). Straightening and repeated dredging also often leads to these channels being incised, reducing risk of flooding damaging crops. The reduction in habitat heterogeneity that this creates reduces instream species resilience (Negishi et al., 2002) and disrupts natural processes. By disrupting natural process, straightening and incising river channels can lead to bank erosion, which is usually accepted due to the relatively low value of land.

Urban rivers are surrounded by much more economically valuable land, which need to be protected. This tends to result in urban rivers being channelised, over-widened and with reinforced banks. They also tend to be straight to speed up the flow of water through the city, improving drainage. Marginal land is also often developed, with impermeable surfaces, which inhibits bank erosion and flooding, fundamentally changing riverine functions (Pilcher et al., 2004, Brookes, 1994). Channelising rivers impacts most species, especially as riparian vegetation is often totally removed for fear of increasing flood risk. However, it is yet unclear how the density and richness of invertebrates are impacted by urbanisation (Krajenbrink et al., 2019), possibly as a result of rural analogues often also being significantly altered from ideal ecological health (Downs and Gregory, 2014).

There is a large body of work on chemical quality of river water and, particularly the impact of sewage treatment works which elevate phosphorus concentrations (Roberts and Cooper, 2018). Another major source of phosphorus is diffuse agricultural pollution (Naden et al., 2016). Elevated phosphorus is significant in rivers as it can lead to eutrophication. Despite the significance of chemical water quality, and the large array of pollutants that occur, there is not a typology of river chemical condition.

One of the most problematic pollutants in rivers is elevated fine sediment loads. Land use changes have resulted in large increases in sediment run off throughout the Anthropocene (Boardman, 2003, Howden et al., 2013), leading to Macklin et al. (2014) coining the term "anthropogenic alluvium". Fine sediment causes multiple issues for sediment sensitive species of animal (Petts, 1984) and macroinvertebrates in particular have been shown to rapidly respond to change (Extence et al., 1999). Large sediment inputs from land use change will result in a change to Lane's balance (Lane, 1955). As the balance attempts to reach quasi-equilibrium, cycles of erosion and deposition will occur longitudinally downstream impacting streams and their ecology for many years to come (Cluer and Thorne, 2014). This has occurred regularly enough that there is a whole body of literature limited to incised streams (e.g. (Bigelow et al., 2016).

3.3. River management and restoration

River management has traditionally altered and removed vegetation and changed channel capacity and form, in favour of using concrete and hard engineering to reinforce river channels for improved drainage and increased flow conveyance (Downs and Gregory, 2014). However, these changes have not been sustainable and, arguably, have not been successful given river hazards these efforts aimed to reduce are more problematic now than ever before (Johnson et al. 2019). Degradation of river ecology can also have unforeseen impacts. For example, macrophytes store and support sediment and, when removed, bed aggradation can be instigated with associated increases in flood risk in the long term (Brooks and Brierley, 1997). Similarly, the loss of native species and/or invasion of non-native species can lead to alterations in river form and dynamics (Greenwood et al., 2018). Beavers are a prime example of how the removal of native species can reduce ecosystem services. Beaver dams slow the flow during flooding, as well as providing a zone of deposition. However, because the dams are "leaky", some sediment enters the water column downstream and therefore there is no rapid erosion (Pollock et al., 2014). Loss of the native organisms can lead to reductions in river resilience, especially in the context of climate change, because river channel form will be in disequilibrium with the surrounding biome (Johnson et al., 2019).

River restoration is a response to past management, which attempts to better work with nature for more sustainable solutions to pervasive problems. It also takes a holistic view of the river (Downs and Gregory, 2014). Restoration works on the assumption that improving part of the system and natural processes will help repair the ecosystem and improve all aspects of it (Palmer et al., 2010). By improving habitat and the physical biotype it is assumed that the ecological, chemical and hydrological feedback processes will also occur (Newson and Newson, 2000). However, serious questions remain about whether this is the case (Palmer et al. 2010). The holistic restoration management is also visible in environmental flows, which aim to restore flow regimes to a more natural cycle (Poff et al., 2010), often incorporating a “flood pulse” (Junk et al., 1989). This approach has come under scrutiny for the degree with which it has simplified a continuum of environmentally important flows (Tockner et al., 2000). The considerable volume of different “flood pulses” needed exemplifies how, although the system is cyclical, it is also very specific and therefore improving one factor does not necessitate the others changing.

One of the major drivers in the increase in river restoration has been the European Union WFD, which attempts to bring all EU freshwaters to a ‘good’ environmental standard (Kallis and Butler, 2001). To assess the performance of freshwater bodies about WFD targets, the UK EA performs substantial monitoring of rivers in England and Wales. The WFD assesses the health of rivers holistically, using four variables; Biological, Physio-chemical, Hydrological and Physical (geomorphological) to contribute to an overall score. WFD research has helped cement the management concept that improving physical habitat is linked to biodiversity (Newson and Large, 2006) thus showing the holistic nature of its design. Beyond the holistic context of these categories the indices used in the biological monitoring are long term indicators of hydrologic and chemical variables and therefore it would be expected that they produce similar results.

3.4. Anthropocene rivers

The majority of global ecosystems have been modified in the Anthropocene (Goudie, 2016), which includes all river catchments in England (Brown et al., 2017). Dufour and Piegay (2009) discuss the way Science looks for processes with and without human influence. However, it appears that all Science occurs in an anthropogenically influenced World and, therefore, we need to start seeing the World with a new perspective due to complete global change (Caldwell et al., 2012).

There has been work suggesting we should manage rivers for the biome they are in rather than the traditional physical based classification systems (Kondolf et al., 2001). However, 75% of Worlds' terrestrial biomes have been altered (Ellis et al., 2010) and, where land use has been altered, so will the river system (Brierley and Fryirs, 2013). These irreversibly altered systems are termed 'Anthromes' (Ellis, 2011) and may correlate to the irreversibility seen in river systems (Dufour and Piégay, 2009). Terrestrially our understanding of these systems is far inferior to our understanding of the bio-physical processes alone (Ellis et al., 2010). Terrestrially, conservation methods do not map onto the processes that create anthromes but, instead map on to the desired processes that may have occurred in the past (Martin et al., 2014). Therefore, anthromes could be a useful tool in developing classification systems for anthropogenic rivers, as well as biogeographical frameworks (Johnson et al., 2019).

Anthromes are a similar concept to emerging and novel ecosystems, where species are found in combinations and abundances not previously seen (Hobbs et al., 2006). This supports the idea that anthromes and human modification do not necessarily correlate to lower biodiversity (Chazdon et al., 2009). The fact that these novel ecosystems can be healthy has resulted in significant revisions to conservation and restoration norms (Hobbs et al., 2009). Due to the lack of systems to draw natural analogues from (Acreman et al., 2014), novel ecosystems are a response to the wasting of precious resources on what may be hopeless quests to "fix" ecosystems. Theses attempts to naturalise already heavily modified systems are not always desired by society or positive in their impacts on biodiversity (Dufour and Piégay, 2009).

Continued management and conservation of anthropogenic streams requires a management framework, which incorporates chemical, biological and physical systems across the spectrum of varying degrees of alteration (Hobbs et al., 2014). Such a framework would provide a fuller set of options for how and when to intervene and to inform how to use limited resources more effectively to achieve management goals (Hobbs et al., 2014). This study will attempt to provide such a river typology for stream types present in the spectrum of human modifications found in the Midlands.

3.5. River classification systems

Early river classification systems can be described as either within river classification systems that describe longitudinal changes (Vannote et al., 1980, Schumm, 1977) or between river classification systems (Holmes et al., 1998). Between river classification systems in particular, are regularly studied and advanced whilst seminal papers form the conceptual basis of this field (Leopold and Wolman, 1957).

Most of these past river classification systems focus on river process, such as sediment transport and hydrological variability, with an implied focus on natural, equilibrium channels. However, it has recently been noted that the streams used to develop these typologies are far from natural, and are already heavily altered by human activity (Montgomery, 2008). Hence, many of the driving and boundary variables that are needed to classify a river system (Thorne, 1997) are anthropogenically altered or not fully appreciated. Many of these classification systems also inherently rely upon an understanding of hydraulic geometry (Wolman and Miller, 1960) yet, hydraulic geometry itself was developed on, and describes, single-thread streams that were altered with mill dams and channelisation (Walter and Merritts, 2008). Despite their limitations, early classification systems formed a basis for study and allowed quantification of how rivers work and the types of streams that exist.

Modern classification systems and typologies started with the Rosgen classification system (Rosgen, 1994), the publication of which led to repercussions throughout the field of river science. This was the first hierarchical framework to take multiple variables into account to typify river classes. However, it suffered from the assumption that river banks are stable and relatively unchanging (Simon et al., 2007). It also did not take climate into account and therefore, was prone to being used to predict river types of the incorrect type, which led to high profile failures of river management schemes where sudden and dramatic channel change took place after channels were constructed using the Rosgen methodology (Kondolf et al., 2001).

By placing variables in a hierarchy, Rosgen's classification initiated the next wave of hierarchical river classification systems, which attributed some variables more explanatory power than others. Rosgen primarily used physical channel descriptors to create the 7 channel classifications and then further delineated these by sediment size (Rosgen, 1994). The difference between these variables suggested that some have greater influence over defining a river system. Rosgen (1994) did not include hydrological variables in the assumption that proxies in the physical data correlate to hydrology (Poff and Ward, 1990). However, this ignores the complexity of hydrological variables (Olden and Poff, 2003) and assumes channels are in equilibrium with the hydrological context. Montgomery and Buffington (1998) created a classification system that focused on geomorphic process not just the current visible channel form in response to these drawbacks in the Rosgen method. It is able to solve these climate based defects by assessing channel condition and response potential utilising basic energy and mass balance equations (Montgomery and Buffington, 1998). While this method is again hierarchical and provides options for enhancing it to relate to other areas beyond the mountains it was designed for it is still relatively constrained in its scope.

RS was developed in Australia as an open-ended classification system, which has the ability to be continuously expanded with new "river styles" that are found to not fit established norms (Brierley and Fryirs, 2000). There remains some form of hierarchy in descriptive variables; for example, valley confinement needs to be accounted for before any other variables when determining river type (Brierley and Fryirs, 2000). Moreover, the method is much more labour intensive than previous classification systems with extensive desk studies and field studies need to be carried out in unison to collect as much information on river type and geomorphic process as possible (Brierley and Fryirs, 2013).

3.6. Relative significance of morphometry in river typologies

All the classification systems currently discussed focus on the physical and geomorphic characteristics of rivers. Classification systems tend to lack a broader focus on other important characteristics, that may also be significant in determining river types and in order to gain a holistic view of rivers. For example, vegetation can define a system through its primary productivity, stabilisation of channel banks and beds, control on flow hydraulics, and alteration of water chemistry. Yet, vegetation has not yet been incorporated into a river typological classification, although it is worth noting that early classification systems used macrophytes to measure the ecological condition of river types (Holmes et al., 1998). The work of Holmes et al. (1998) suggests that vegetation can be a useful input parameter into classification tools because of their stability in channels, especially if not impacted by anthropogenic activity (Holmes et al., 1998).

Stream water chemistry and biology have been even less studied in the context of creating a typology of streams. It is far more common to partition rivers based on their condition i.e. good or bad (Lepom et al., 2009). In the past this may have been because water chemistry is not immediately visible and is relatively hard to measure. However, there is an increased demand for understanding the wider context for river water chemistry and pollution, which now incorporates biomonitoring and citizen science monitoring programmes, such as the Riverfly Partnership (Brooks et al., 2019). The various variables used in river surveys, such as the River Habitat Survey (RHS; (Raven et al., 1998) and Stream Reconnaissance survey (SRS; (Thorne, 1998), as well as macroinvertebrate monitoring, could be used to dictate classification systems as they are a direct insight into the variables that river managers consider important in defining streams that are degraded.

3.7. Catchment context for classification systems

It is important to understand the role that the whole catchment plays in controlling river type, as highlighted in the RS framework (Brierley and Fryirs, 2013). Despite this, land use at the catchment scale plays little role in the study of RS, unless the modified method is selected (Brierley and Fryirs, 2013). In fact, the normal RS method only includes percentage forest cover as a catchment scale indicator (Brierley and Fryirs, 2000). However, in Europe all streams have been impacted by land use change to some degree (Montgomery, 2008) and, therefore it is inappropriate to work off a classification system predominantly designed for "natural ecosystems" that does not take proportional catchment scale land use into account.
Land use change has been linked to increases in sedimentation (Trimble, 2009), leading to the creation of classification systems working on the basis that sediment balances change river type (Notebaert et al., 2018). Some of these changes are considered to be irreversible if the sediment supply has changed enough to cross a threshold (Notebaert et al., 2018). The viability of a threshold over a continuous scale has been discussed in regards to classification systems (Thorne, 1997). However, there has been little consideration of how stream types at impacted sites are able to return to previous states, with the notable exception of the work of Montgomery and Buffington's (1998) classification that incorporates stream resilience to change. However, this classification still only focuses on physical changes (Notebaert et al., 2018) and it would be especially useful to understand how the threshold concept may operate in terms of ecological and chemical function.

Notebaert et al. (2018) looked at rivers that were expected to have been impacted by anthropogenic activity, which is similar to the modified RS. Davenport et al. (2018) extended past work by constraining site selection to urbanised UK streams, improving statistical classifications. Statistical methods are useful typifying tools if attempting to identify new channel types that are not identified when using more subjective methods (Buffington and Montgomery, 2013). Variable hierarchy is not totally removed from Davenport et al. (2004) study because the selection of variables is foremost defined by their feasibility to be included in the large input UHS. Secondly, to format data for analysis it was transformed to create an index between 1 and 10, with semi-quantitative attribute values, thus artificially increasing and decreasing the relative value of some variables.

Regardless, none of these classification systems have fully taken into account that the whole system is anthropogenically impacted and, therefore, all streams in the UK are modified to some degree. Instead, streams represent a continuum of anthropogenic impact (e.g. Johnson et al. 2019) and classification systems would usefully be improved to specifically account for rivers as they now exist in the Anthropocene, with less inherent significance given to natural processes. The Midlands of England therefore represents a useful study area given its long history of industry, mining, urbanisation and land use change.

4. Methodology

4.1. Methodological overview

The development of different classification systems have used a range of input data, with a variable quantity of examples and time-steps used to generate clusters, which depends on resources and data demands (Cochero et al., 2016, Davenport et al., 2004). There is also a division between studies using solely statistical tests based on a few variables and those that collate historical and desk study data to provide a broader view on river type but with less statistical analysis (Brierley and Fryirs, 2000).

This study was carried out using cluster analysis, which required a data set with considerably more cases than variables (Revelle, 1979). To counter time constraints, 50 midlands rivers were chosen (Fig. 1.) with 36 variables used and split into 4 different categories, providing significant power for cluster quantification (Borcard et al., 2011). It also provides a more extensive background dataset than most other studies of this sort. Much of the data used were available as secondary data but, due to the study focus on relatively small streams and the paucity of high-resolution topographic data, primary data collection was required for some variables (e.g. slope).
A statistical methodology, detailed below, was then used to quantify clusters of river sites and to explore the controlling variables on their grouping.

4.2. Site selection

Sites were chosen based on a range of factors. Firstly, they needed to be of approximately similar size to avoid incorporating site variability associated with river size. Here, all rivers were approximately 8-12 m wide and relatively upstream in their catchments. Only one site on each river was used to ensure all sites were independent and each site was selected so that it was within 2 km of a National River Flow Archive (NRFA) gauging station, so that flow measurements were available. Within these constraints, sites were chosen within the Midlands of England so that a range of land-uses and locations were attained but keeping them within the same approximately climate zone. NRFA data was also used to ensure that the average flow was safe for fieldwork and, in particular, for sediment grain size sampling to be plausible. Sites also needed to have EA water chemistry and biological monitoring locations within 2 km for secondary data assessment. The 50 resultant sites that met all these criteria are shown in Figure. 1.

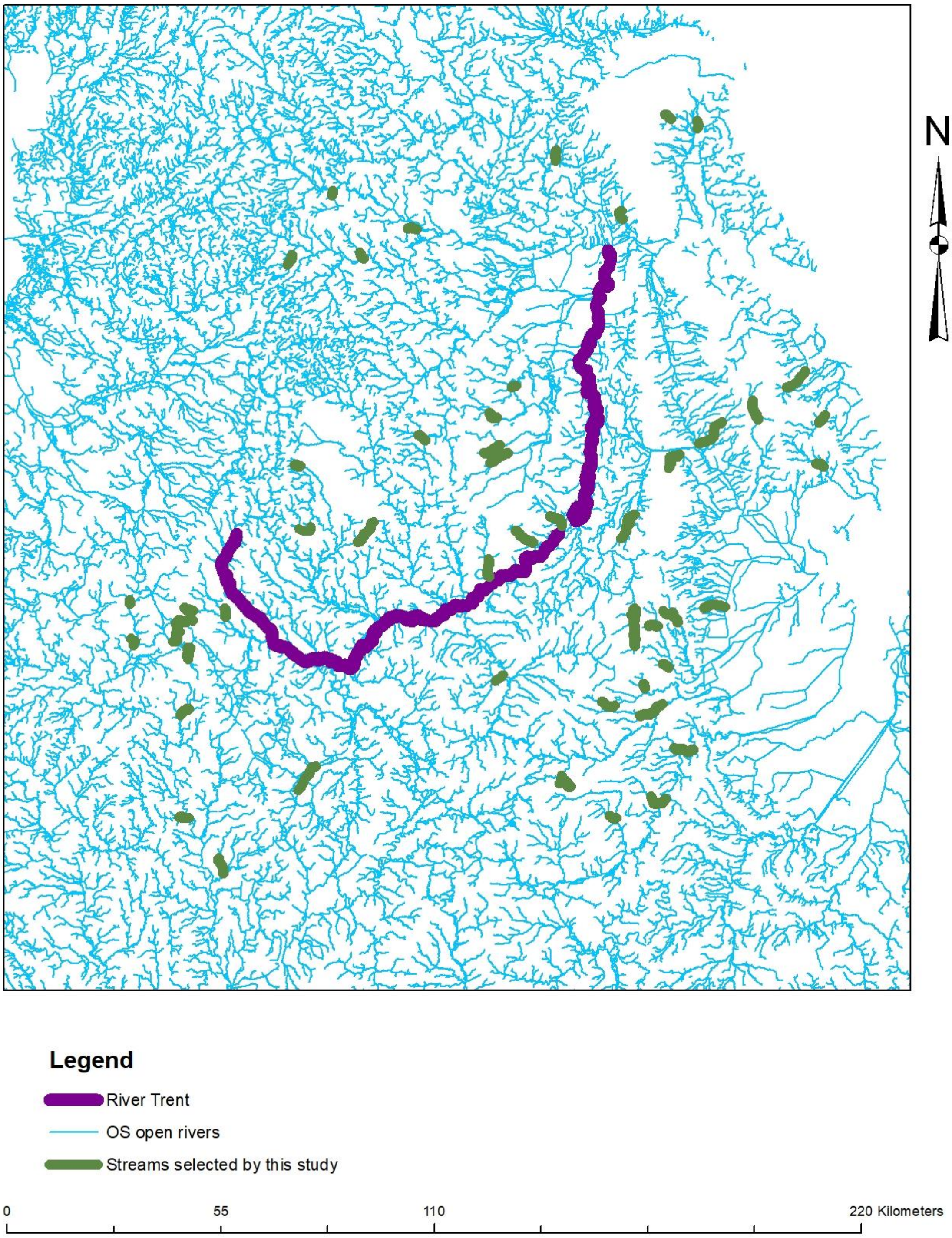


Figure. 1: 50 river reaches chosen for fieldwork, based on proximity to gauges, EA sampling and size. Green highlights are the chosen streams, light blue are OS river map channels to display chosen sites in context. The purple channel is the River Trent. While not all rivers in the study are in the Trent catchment it allowed for centring of the sites.

4.3. Assessment of characterising variables

Data collected for the cluster analysis needed to be quantitative or semi-quantitative (Borcard et al., 2011). As river systems are highly variable, there is a substantial range of potential metrics that could be used for cluster analysis. For example, Olden and Poff (2003) documented 171 different parameters that can be used to document flood hydrographs. To avoid redundancy and cross-correlation in variables, variables were carefully selected for this study. Following the WFD, these variables were partitioned into one of four categories (Kallis and Butler, 2001): Biological, Chemical, Hydrological and Physical. Firstly, the variables used on other river typologies was collated. The typologies studied were the Stream Reconnaissance Survey (SRS) (Thorne, 1998), River Habitat Survey (RHS) (Fox et al., 1998, Raven et al., 1998), UHS (Davenport et al., 2004), RS (Brierley and Fryirs, 2000), the Rosgen classification (Rosgen, 1994), WFD requirements, and additional anthropogenic geomorphic variables that may be important (Goudie, 2016).

This resulted in a list of several hundred variables, with much repetition. Once obvious repeats were removed, the list was reduced to approximately 50 variables, from which 36 variables were chosen to be taken forward in this study based on their significance to river form, their plausibility of measurement, and to ensure a roughly equal number of variables were selected in each of the four assessment categories. These variables are documented in table 2 and explained below.

Table. 2: The 36 variables which were included in the study. These variables are split into four categories based on the WFD classifications and constituted 4 separate hierarchical cluster analyses. The Suitability of this typology was be comparable against WFD classifications and Heavily Modified Water Body status.

Biological	Chemical	Hydrological	Physical
WHPT N Taxa	Orthophosphate, reactive as P	Q50	Depth
WHPT ASPT	Sewage (population equivalent)	Q10	Channel Slope
Life Family Index	Nitrogen, Total Oxidised as N	BFI HOST (Soil hydrology)	Habitat Modification Score

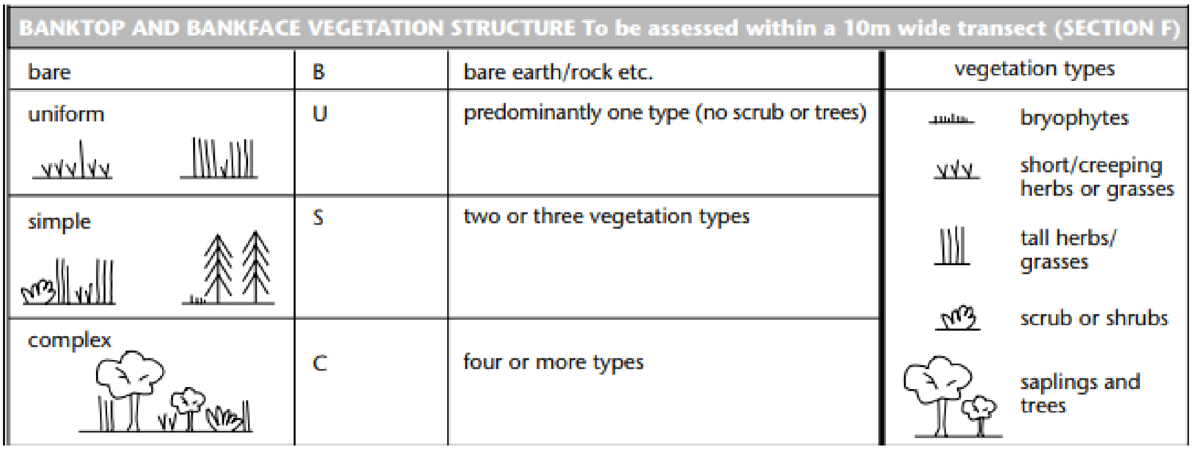
PSI Family Score	Dissolved O_2	DPS BAR (Catchment Slope)	Sinuosity index
Riparian vegetation complexity	Upstream roads	Catchment Area	D50
	PH	Elevation (max)	D95
		Flow Characteristics	D84
			Urban % land use
			Arable % land use
			Woodland % land use
			Grassland % land use
			Other % land use
			Visible bank erosion
			Anthropogenic litter

4.3.1. Biological variables

Biology holds executive power in the WFD (Borja et al., 2006) and is intrinsically linked to a healthy river (Norris and Thoms, 1999). Biology also exerts a control on river morphology by modulating Lanes' balance (Lane, 1955, Moore, 2006), including altering the flow required to entrain sediment (Johnson et al., 2011) and protecting and supporting bank materials (Anderson et al., 2004). Moreover, ecology has an intrinsic right to exist (Sterling et al., 2018).

Riparian vegetation

As primary producers, riparian vegetation provides the raw material for riverine food webs (Vannote et al., 1980) and is likely to contribute to physical morphology of rivers due to its impact on bank stability (Simon and Collison, 2002, Thorne, 1990). Furthermore, riparian shade exerts a control on river temperature (Johnson and Wilby, 2015), which is of significance to river water chemistry. Vegetation complexity was recorded following the methodology of the RHS, shown in Fig. 2.



BANKTOP AND BANKFACE VEGETATION STRUCTURE To be assessed within a 10m wide transect (SECTION F)			
bare	B	bare earth/rock etc.	vegetation types
uniform	U	predominantly one type (no scrub or trees)	bryophytes short/creeping herbs or grasses
simple	S	two or three vegetation types	tall herbs/ grasses scrub or shrubs
complex	C	four or more types	saplings and trees

Figure. 2: RHS options available for semi-qualitative measurement of riparian vegetation. Coded 1-4, with 4 being complex vegetation, in the cluster analysis.

WHPT N Taxa

Macroinvertebrates are commonly used to assess water quality and ecosystem health in rivers (Guareschi et al., 2017). Furthermore, they are central to the lotic ecosystem as a food source to higher trophic organisms and because of their role in decomposing organic matter (Callisto et al., 2001). Taxa abundance objectively measures macroinvertebrates without attributing scores to species. Therefore, unexpected functioning ecosystems can be analysed for density of fauna (Acreman et al., 2014). Here, the number of taxa that score in the WHPT biomonitoring score were assessed using secondary macroinvertebrate community data collected by the EA. Data was collected using a 3-minute kick sampling and 1-minute hand search, which are national standards (Everall et al., 2017).

WHPT ASPT

The WHPT biomonitoring index scores macroinvertebrates based on their resilience to pollution (Paisley et al., 2014), replacing earlier biomonitoring scores for assessment as part of the WFD (UKTAG, 2014). The WHPT is specifically adapted to account for invasive species (Guareschi and Wood, 2019) and, using the Average Score Per Taxa (ASPT), allows a direct comparison between sites, regardless of their relative community abundance (Armitage et al., 1983). The WHPT ASPT is calculated as: WHPT ASPT = Sum AB / WHPT N TAXA. Where AB = value for each taxon based on its abundance and resistance (UKTAG, 2014)

LIFE Family Index

The Lotic-invertebrate Index for Flow Evaluation (LIFE) was developed in the UK (Extence et al., 1999), using macroinvertebrates to infer information about the hydraulic regime. Certain taxa are adapted to different flow velocities and so are found in higher abundance in particular flow environments (Miller and Golladay, 1996). The LIFE score is calculated as follows, using the same scoring taxa as the WHPT: Taxa are separated into categories created from literature based on their flow tolerances and these are attributed a score based on the abundance of taxa in each category (Table. 3.). Scores are combined to give the LIFE Family Index.

Table 3: The Lotic-invertebrate scores Index is made up from adding together scores found in this table. Flow groups are based on literature behind individual taxon (Extence et al., 1999).

Flow Groups	**Abundance of invertebrates**			
	1-9	10-99	100-999	1000+
1 rapid	9	10	11	12
2 moderate/fast	8	9	10	11
3 Slow/Sluggish	7	7	7	7
4 flowing/ standing	6	5	4	3
5 Standing	5	4	3	2
6 Drought resistant	4	3	2	1

PSI Family index

The Percentage Sediment Sensitive (PSI) family index measures the percentage of the macroinvertebrate community that are adapted to streams with high fine sediment content (Extence et al., 2013). The measure correlates strongly with both deposited and suspended sediments (Turley et al., 2014). Macroinvertebrates are known to be negatively affected by fine sediment (Jones et al., 2012), with changes in community indicative of habitat degradation. The PSI is calculated similar to LIFE scores by attributing a level of sediment sensitivity to different taxon and then adding up the various abundances of different groups and attributing an overall score (Table. 4.)

Table. 4: Table to Calculate PSI scores. All invertebrates are placed in a fine sediment sensitivity rating and thereafter abundance is calculated to give a combined site score (A. Extence et al., 2013).

Fine Sediment sensitivity rating	Abundance of taxa			
	1-9	10-99	100-999	1000+
Highly sensitive	2	3	4	5
Moderately sensitive	1	2	3	4
Moderately insensitive	1	2	3	4
Highly insensitive	2	3	4	5

4.3.2. Chemical variables

Anthropogenic chemical pollution has led to unacceptable regional changes (Steffen et al., 2015) and has a direct impact on humans (Lepom et al., 2009), as well as wildlife. Over half of the streams in Europe are toxically polluted (Malaj et al., 2014). Ecosystems are also suffering from physio-chemical change (Reyjol et al., 2014) and the mix of different chemicals is often underappreciated for the damage it can cause (Carvalho et al., 2011). The EA collect chemical data monthly from spot measurement sites across England and Wales. Their monitoring is non-randomised (Nixon, 1996) but, the method is designed to be defendable in court (Allan et al., 2006). Here, monthly values were averaged between the year 2000 and 2018 to obtain a long-term average for each site. As much as possible was done to remove error including removing a case with a Phosphorus value outlier an order of magnitude different to the other values. Specific parameters used were:

Mean annual temperature

Temperature is rarely included in classification systems but, can explain bio-chemical responses (Thomson et al., 2004) because it is one of the most important variables controlling lotic habitat (Poff and Ward, 1990). Spatial (Frechette et al., 2018) and temporal (Wilby et al., 2014) change in stream thermal regime are known to be vital in delineating the survival and stress of aquatic species'. Here, monthly stream temperature from the 18-year record was used to generate a mean value. Because the streams used were all relatively small, they are likely more responsive to changes in climate and shade provision than larger lowland streams (Johnson and Wilby, 2015).

Phosphorus

Phosphorus (P), Nitrogen (N), Carbon dioxide and light are the base materials for primary productivity (Jarvie et al., 2018). P is the limiting factor for growth in many aquatic systems (Mallin et al., 2006) and so communities are composed around a “natural” baseline (Smith, 2003). Where anthropogenic activity increases P, the competitive balance of ecosystems are altered from its base upwards (Mainstone and Parr, 2002), ultimately causing eutrophication (Civan et al., 2018). Here, an 18-year mean value from EA orthophosphate measurements was used. Orthophosphate represents the biologically available component of the P load and, therefore, has more direct significance to aquatic life.

Sewage treatment population equivalent

Sewage Treatment Works (STW) can contribute a large proportion of a rivers’ discharge (Shafer et al., 2009) and usually represent point sources. STW can produce more anthropogenic P than can be buffered by sediment (Roberts and Cooper, 2018) and can be the main control of P levels in urbanised catchments (Bowes et al., 2018). Some pollutants from STW have no natural parallel, such as pharmaceuticals which can have extensive impacts (Jelic et al., 2011). Moreover, *E. Coli* is removed during treatment but combined sewage overflow systems can release raw effluent during flooding (Gracia, 2018). Population equivalent was used as a proxy for the size of STW, and was collated from the European Commission urban wastewater website (ECUWW, 2019) and cross-checked with the European Environment Agency (EEA, 2019). Where there was a discrepancy, the conservative value was taken and these were then coded into no STW [0]; small town STW up to 10,000 population equivalent (Doxiadis, 1968) [1], and; large town STW [2].

Nitrogen

Like P, N is a limiting factor and required for plant growth. N is often found in unacceptable concentrations in rivers due to the application of agricultural fertilizer and their run-off into waterways (Wang et al., 2018). As with P, an 18-year mean value was calculated using EA Total Nitrogen measurements, because of all the forms of chemical nitrogen in river water, this had the most complete measurement record by the EA.

Dissolved oxygen

Dissolved oxygen is determined by the balance between reaeration at the water surface and oxygen uptake within the stream (Williams et al., 2000). Modified streams often have more homogenous flows (Serlet et al., 2018), reducing aeration. Algal biomass, dissolved organic matter, ammonia and oxygen demand of sediment all remove oxygen (Sánchez et al., 2007). Mean dissolved oxygen was calculated over the 18-year monitoring period.

Upstream roads

Road and vehicle pollution is a pervasive and universal anthropogenic impact on all aspects of the environment (Coffin, 2007) but, there is limited research on its fluvial impact. Sediment input into waterways is increased by road building and vehicle passage (Johnson et al., 2002). Discharge is also altered by road drains and altered run-off (Jones, 2000). Finally, road salt is laid during winter in the UK, which is soluble and toxic to flora and fauna (Godwin et al., 2003). It was not possible to quantify the impact of road density on rivers in this study and, therefore, bridges were used as a proxy. This was compiled through a desk study using EA open data and google maps. Cases were attributed 1 point for every vehicle lane that crossed a river upstream of the sample point, to the source of the stream. Therefore, a standard 3 lane motorway would score 6 points.

pH value

pH has the ability to seriously damage aquatic ecology (Winterbourn et al., 2000). Acidifying rivers is generally caused by anthropogenic activity, including acid rain (Leivestad and Muniz, 1976) and acidic mine water runoff (Oberholster et al., 2017). Mine water rebound results in considerable mine water pumping in the Midlands (Younger and Adams, 1999), which is a potential driver of change despite reed-bed filtration (Falagán et al., 2016). Alternatively, farm liming may lower or buffer pH values (George et al., 2018). Mean pH from EA monitoring over 18-years was used.

4.3.3. Hydrological variables

Kondolf et al. (2001) showed that if the full hydrological cycle was not taken into account, river management can cause serious detriment and may lead to sudden, large changes in river form. Thus, it is vital for a classification system to include hydrology as a key driver of river processes. Moreover, different flows are needed for ecosystem health, including flood flows (Junk et al., 1989). The data, documented below, mostly comes from the NRFA gauging network.

Q10, Q50 and Q95: high, medium and low flow

Base (Q95), median (Q50) and high flows (Q10) were selected to describe the flow regime. Low flows are increasingly important due to climate change (Stott, 2016) and over-abstraction reducing the base flow needed for species survival (Horne et al., 2017). Base-flow is also important in driving connection with groundwater, with implications for river temperature (Dugdale et al., 2013). High flows are integral to understanding the anthropocentric view of rivers and are effective at moving sediment (Wolman and Miller, 1960), making high flows important for geomorphological understanding. The 10^{th}, 50^{th} and 95^{th} percentiles were calculated from NRFA flow gauge daily time stepped data since the year 2000. These are routinely used variables in parameterising flow regime.

BFI HOST soil

The degree of soil permeability controls both the speed of through-flow and the recharge of the catchment aquifer system (Scanlon et al., 2002). Impermeable soils create rapid run off and a flashy hydrograph (Musgrave, 1935). Hydrological models consider soil moisture a vital physical input (Devia et al., 2015) improving predictions by as much as 10% (Laiolo et al., 2016). Urbanisation or agriculture can change soil composition (Doichinova et al., 2006), which would create anthropogenically influenced stream types. The parameter used here is the Base Flow Index: Hydrology of soil types (BFI Host) from the NRFA dataset. 29 soil classes are presented on 1 km gird cells, which are then indexed to create a comparable statistic of soil influence (Boorman et al., 1995).

DPS BAR Slope

Catchment slope influences run off (Agassi et al., 1990) and infiltration, especially where the slope is very steep (Poesen, 1984). Drainage Path Slope (DPS) is the mean of all the slopes in the catchment expressed in metres of slope per kilometre of area. The data profile was designed for the flood estimation handbook (CEH, 2019) displaying the objective speed precipitation would reach the river channel. DPS BAR can be compared to the actual time it takes water to reach the channel due to anthropogenic alterations (Barron and Barr, 2009). Anthropogenic slope change may also be visible from the effects of building and flattening works (Olshansky and Kartez, 1998) making it another indicator of modified stream types.

Catchment Area

Discharge was expected to be directly related to catchment size (Leopold and Maddock, 1953). Discontinuity between these variables suggests other factors are influencing the system, such as water abstraction. Storm drains can also alter this relationship especially in the case of small catchments where towns may span over catchment borders (Jones, 2000). Catchment size was calculated here in ArcGIS, by calculating the upstream contributing area from NRFA gauge locations on 5 m resolution DEMs.

Elevation

Elevation is a proxy for stream location within the catchment (Schumm, 1977) and the stream order (Shreve, 1966) because higher elevation streams are more likely to be first order tributaries in the supply zone. As the River Continuum Concept dictates, physical conditions ecological communities should systematically change with downstream distance along a river (Vannote et al., 1980). Despite its significance, river typologies rarely account for elevation when comparing stream type (Thomson et al., 2004). Here, elevation was taken as the highest elevation point in the catchment from 5 m resolution DEMs, within the catchment area upstream of NRFA gauges.

Flow Characteristics

Flow characteristics are heterogeneous in rivers, creating a variety of hydraulic habitat (Beisel et al., 2000), to the benefit of organisms (Lacey et al., 2012). Multiple flow characteristics can oxygenate water and create refuges whereas, a lack of flow can encourage eutrophication. Here, the semi-quantitative RHS method was used (Raven et al., 1998) to record different discrete flow types. Measurements took place over a 500m reach, within which all flow characteristics occurring in 5m sections were recorded. The flow options were; chute flow, broken standing waves, unbroken standing waves, rippled flow, smooth flow, no perceptible flow, marginal dead water. These were recorded as count data. The presence or absence of perceptible flow was then used as a separate variable as this was considered an important indicator of river type and health.

4.3.4. Physical variables

Physical variables describe the geometric shape of the channel and therefore describe the geomorphic processes that have occurred (Brierley and Fryirs, 2013). Geomorphological aspects considered in the WFD can only change the grading of a stream from Good to High Status (Newson, 2002). However, channel geomorphology at the reach scale is considered to be a vital underpinning of any river classification system due to its controls on flow characteristics and habitat (Benda et al., 2004).

Channel Width and Depth

Channel width and depth are used in most classification systems (Thorne, 1997), as is stream size (e.g. Taube et al., 2019, Rosgen, 1994) used a dimensionless combination of width and depth but, in this study the streams are all small allowing the use of separate variables. Available Lidar at 1 m resolution was insufficient to characterise channel width and so primary field data was collected. Channel depth was measured from the river bed to bank top at the thalweg, resulting in water depth at bankfull (Simon et al., 2016). To account for spatial discrepancies in bank height this value was averaged with two other measurements 15 m upstream and downstream (Williams, 1978). Width was measured between bank tops at each of these three points and also averaged.

Channel slope

Slope is a prerequisite of a river channel (Charlton, 2007) and one of the main drivers of river processes (Lane, 1955). As such it is a primary control of river character and behaviour, delineating class boundaries in RS (Brierley and Fryirs, 2013). Traditional GPS had too great a degree of error and Spectra precision MobileMapper 120 (Trimble Navigation, 2012) was found to be unreliable in the field due to riparian vegetation disrupting signals. Therefore, slope was calculated from surveyed data using a dumpy level to record bed elevation at a minimum of 30 m apart but, usually 60 m apart, which is similar to the 7 channel widths used in some other studies (Stock et al., 2005). Slope was recorded at the thalweg in all cases for standardisation.

Habitat Modification Score

The habitat modification score (HMS) records direct anthropogenic management on water courses. Since rivers are longitudinally connected (Williams and Wolman, 1984) the RHS was adapted so it accounts for de-localised drivers (Raven et al., 1998). Modifications present are attributed a value producing a quantitative ordinal score, which is suitable for cluster analysis. Here, observations were taken at 10 sites over 500m reaches. Every 50 m a score was attributed based on direct management (Table. 5.) adapted from the HMS rules (documented in Appendix. A.). The entire 500 m reach was then also measured for abundance of certain elements (shown in Tables. 6 and 7.) and all these scores were added together for a site total. It should be noted that upstream roads were already analysed as a chemical variable and described above. In the HMS, only roads local to the specific reach are recorded, documenting a different pressure to prior parameters.

Table. 5: Section 1 of Habitat Modification Score adapted for this study. Every 50 m each of these features are looked for. If they are found, they are tallied together and then added to section 2 de-localised features for the overall score.

Spot check feature	**Attributed score**
Reinforcement to banks (not bioengineered)	2
Reinforcement to bed	2
Re-sectioning of bank or bed	2
Two stage bank modification	1
Embankment	1
Culvert	8
Dam or weir, unchanged	2
Dam or weir, adapted to have a fish pass	1

Table. 6: Section 2(a) of Habitat Modification Score system. Features are scored based on their abundance for the whole 500m reach under study. Scores are combined with section 1 and 2(b) for total Habitat Modification Score.

Feature	**Abundance = 1**	**Abundance ≥ 2**
Road bridge	1	2
Enhancements including groynes	1	2
Flow controls present	5	N/A
Partially re-aligned	5	N/A
Straightened	10	N/A

Table. 7: Section 2(b) of Habitat Modification Score system. Features are scored based on the abundance of spot check sites (50m sections of the total reach) in which they are found. Scores are combined with section 1 and 2(a) for overall Habitat Modification Score.

Abundance of spot checks with presence	Score for Poaching	Score for Bioengineering
1-2	1	1
3-5	2	2
6+	3	3

Sinuosity Index

Channel sinuosity can be classified into straight, meandering and braided (Leopold and Wolman, 1957) and creates a continuum when measured by a sinuosity index (Brice, 1964). Anthropogenic actions have directly affected channel sinuosity (Downs and Gregory, 2014), whilst also possibly indirectly controlling it through alterations to catchment land-use (Brown et al., 2018, Montgomery, 2008). Sinuosity was calculated as the ratio of channel length to valley length over a 200 m valley length (Champkin et al., 2018). This was achieved in a desk study using channel centreline data from 1:125,000 scale OS open data map in ArcGIS. Polylines were used to compare centreline to valley width (Sapkale et al., 2016, Leopold and Wolman, 1957). Centreline to thalweg difference was not considered significant enough to affect results.

D50, D84 and D95

Grain size is a delineating variable in most classification systems (Brierley, 1996). To some extent, sediment's ability to delineate streams may be due to it acting as a limiting factor for other processes (Haddadchi et al., 2018) including the requirement of sediment as habitat for many species (Maddock, 1999). To collect substratum size, a Wolman's pebble count (Wolman, 1954) was implemented using a gravel-o-meter (Hey and Thorne, 1983). 100 sediment grains were measured at each site in order to calculate 50, 84 and 95th percentiles (Kondolf, 1997). Operator bias was controlled by the same operator carrying out the procedure at each site and by following a regular grid with the sampled grain being randomly selected as that at the toe of the operator's boot at each point on the sample grid (Wolman, 1954).

Land use Indicators

The RHS and SRS both use multiple land use categories, which do not take the entire catchment into account. RS states the need to manage a catchment as a whole but, does not explicitly include catchment scale land use beyond forested area unless using the modified system (Brierley and Fryirs, 2013). NRFA provide land cover data in the form of percentages of total catchment area upstream of gauged stations. The land cover data used fell into 5 categories; Woodland, grassland, arable, urban and other. These data were made up from a combination of Land cover map 2000 and Land cover map 2007.

Visible Bank Erosion

Bank erosion allows a degree of freedom in which a river can naturally adapt (Kondolf et al., 2001). Thus some channel erosion is natural and desirable (Florsheim et al., 2008). However, large amounts of bank erosion can cause damage (Couper and Maddock, 2001), erode ecological habitat (Naiman et al., 2010) and, affect downstream sediment supply (Schumm et al., 1984). This suggests that large scale bank erosion may be symptomatic of a degraded system (Brierley and Fryirs, 2013). In order to record bank erosion, a visual assessment of banks was made over a 50 m length of river at each site, with sites partitioned into one of five categories; no erosion [0] small erosion on 1 bank [1] small erosion on both banks [2] heavy erosion on 1 bank [3] and heavy erosion on both banks [4].

Anthropogenic litter

Litter counts gave a measure of how much direct human influence each site was receiving and, in some cases, litter appeared to change stream functioning. Firstly, some streams had large numbers of bricks and large litter that was orders of magnitude larger than the sediment size. Secondly, anthropogenic litter breaks down within a stream and can cause pollution. To standardise recordings, a 5 m plot was marked out at each site and all litter was counted within the area. Notes were also made of the main types of litter for further analysis but, in cluster analysis the total number of litter pieces was used.

4.4. Statistical methods

All statistical methods were carried out in R. Primarily this involved the application of cluster analysis using the “vegan” package to determine clusters based on the dissimilarity of variables. The dissimilarity matrix used to prepare the data for cluster analysis was “Gower” as this allowed continuous and categorical variables to be compared once they had been converted to numerical values. The dissimilarity matrices of different categories of data (e.g. hydrological, physical, chemical and biological) were compared in scatter graphs and tested with Pearson's regression.

Wards clustering algorithm was used in cluster analysis, as used in previous studies of river typologies (Davenport et al., 2004). This is a hierarchical clustering mechanism and therefore separates streams into clusters to reduce within group variance until all individual cases are separated (Borcard et al., 2011). Cross-clustering (Zhou et al., 2019) was attempted to mitigate the reduced power from removed variables, but it was considered too weak to accurately predict values in this varied data set.

Silhouette widths were used to define the number of clusters analysed for each category. These represent within group mean intensity, which is the number of clusters appropriate so that the average distance between a case and the other cases in a cluster is reduced as much as possible (Borcard et al., 2011). If a cluster number had been selected with lower average silhouette widths, it would produce clusters where the cases had a lower degree of membership to the cluster and therefore lack the internal homogeneity needed to define clusters (Griffith and Amrhein, 1997). Although this was done statistically, having a choice in the method used results in the choice of clusters being somewhat subjective (Davenport et al., 2004) although arguably less so than pre-defining them in the context of what characteristics were expected to delineate into clear clusters.

Dendrograms showing the results were drawn using the "ggplot2" package. Non-metric multidimensional scaling which would provide another opportunity to visualise how these clusters sit in multidimensional space. However, due to the number of variables and the variation in the system these could only display a small percentage of the results on the two axis and therefore they did not produce the desired visual benefits (Appendix. B.).
All statistical tests were carried out using the "car" package. These included students t tests after the assumptions of normality and homogeneity of variance had been tested using Shapiro-Wilks and Levene's tests, respectively.

To create the presence absence graphs, a matrix was produced by the cluster output and multiplied against other category outputs.

5. Results

5.1. Relationships between environmental variables

In order to check for autocorrelation, Pearsons correlations between all continuous variables used in subsequent analysis were tested (Fig. 3.). This information is important to appropriately interpret later statistical results. Some significant correlations exist but these have relative weak explanatory power ($R < 0.6$), with the exceptions the D50, D84 and D95, which are all measures of grain size thus correlating strongly with each other, and the Q10, Q50 and Q95, which are all measures of discharge and so again correlate together. Grassland and arable land are also highly correlated. The lack of strong correlations in certain variables is also interesting, for example, the macroinvertebrate biomonitoring indices do not correlate with the measures they are proxies of, such as the LIFE scores and discharge (Q) and PSI score and grain size (D).

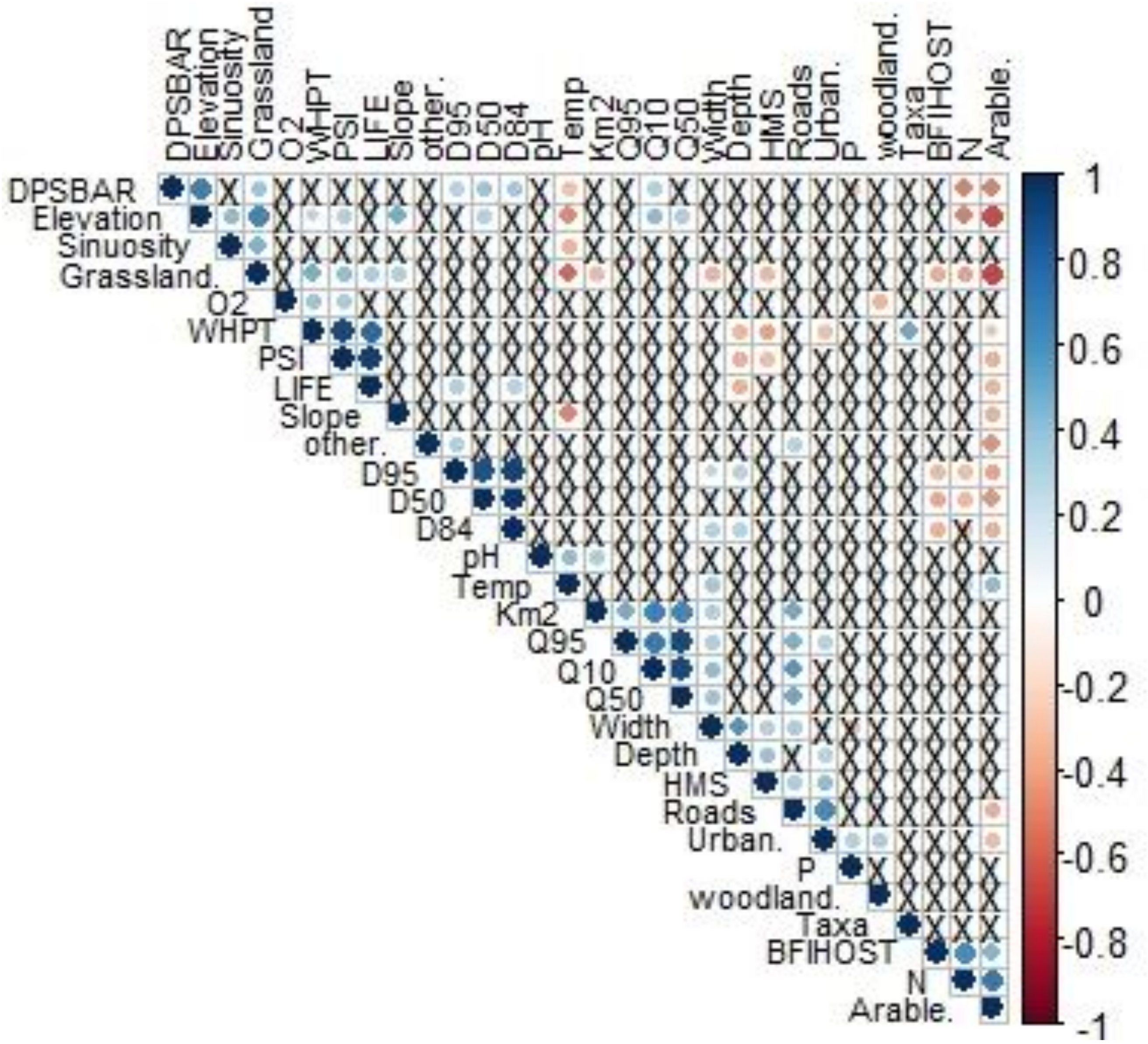


Figure. 3: Correlation matrix for all continuous variables used in the study. Blue indicates positive correlation, red indicates negative correlation, size and brightness indicate the strength of this correlation. All the crossed out cells were not found to be significant at the 0.05 threshold using spearman's rank test on R.

5.2. Cluster dendrogram of rivers

5.2.1. Overview clustering

The clustering algorithm provided different clusters compositions in each of the four categories i.e. biological, chemical, morphological and hydrological (Fig. 4.). It is clear that the four cases do not have the same grouping of rivers and rivers are do not obviously align in groups spanning all 4 categories, which will be explored in more detail in Section 5.3. The dendrograms shown in Figure 4 present a visual representation of the clusters.

Dendrograms were cut according to highest silhouette width to present the groupings with the highest internal homogeneity (Borcard et al., 2011), resulting in a lower number of more clear categories. Silhouette width diagrams (Appendix. C.) Visualise the grouping options including the preferable number of clusters. Each of these four categories of cluster analysis is expanded upon in the following sections.

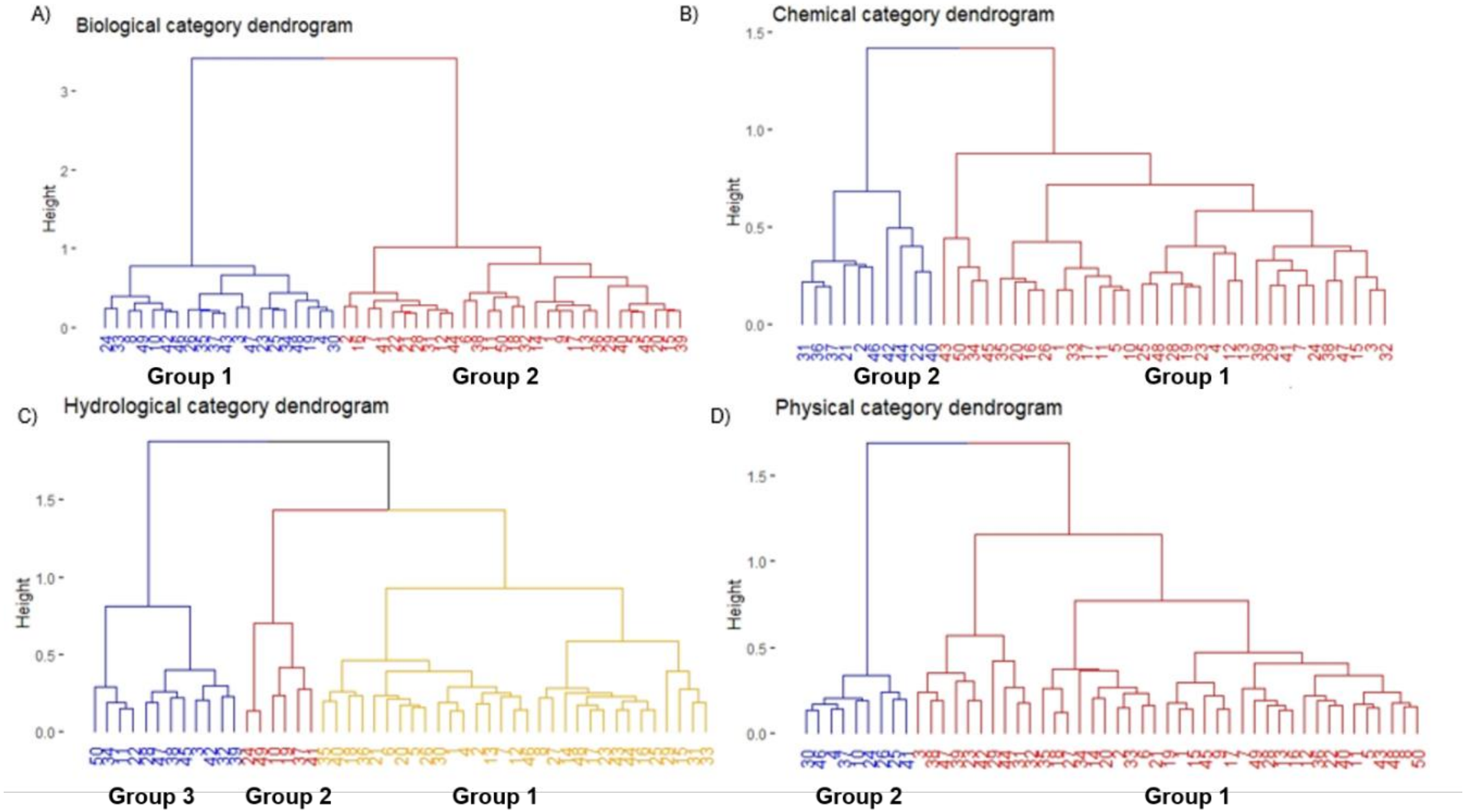


Figure. 4: Dendrogram clusters made up with the different categorical variables. The decision on where to cut the trees for groups was based on average silhouette width with the point of most explanation being chosen as the most descriptive grouping system. The case numbers refer to the same rivers in each example.

5.2.2. Biological clusters

Cluster analysis for the biological conditions produced two distinct clusters, with an average silhouette width of 0.274. Group 2 has significantly higher LIFE, PSI and WHPT values (Table. 8.). It also has a higher median for the degree to which the vegetation was considered complex, with the exception of one outlier (Fig. 5.). Surprisingly, the number of taxa is only marginally significantly higher in Group 2 (alpha 0.1, despite the variables representing better ecology.

Table. 8: Comparison of Biological clusters finding them to have created significantly different ecologically healthy and ecologically unhealthy clusters. It should be noted that N taxa is not significantly different but the index valuations of these taxa is. As a categorical variable riparian vegetation does not produce a true mean and this should be taken into account when considering the significance of this difference. Significance was calculated using a paired t-test in R with alpha value of 0.05. All categories were first checked for normality with a QQplot and Shapiro-Wilk normality test.

Measure	Group 1	Group 2	Significance value
N	20	29	
Riparian Vegetation **	2.45	3.14	0.003*
Riparian Vegetation Median	2	3	
PSI Family Index	30.40	59.86	<0.001*
LIFE Family Index	6.43	7.41	<0.001*
WHPT ASPT	4.43	5.65	<0.001*
WHPT N TAXA	20.33	23.41	0.082

* denotes significant at alpha 0.05

** denotes categorical variable therefore not a true mean average

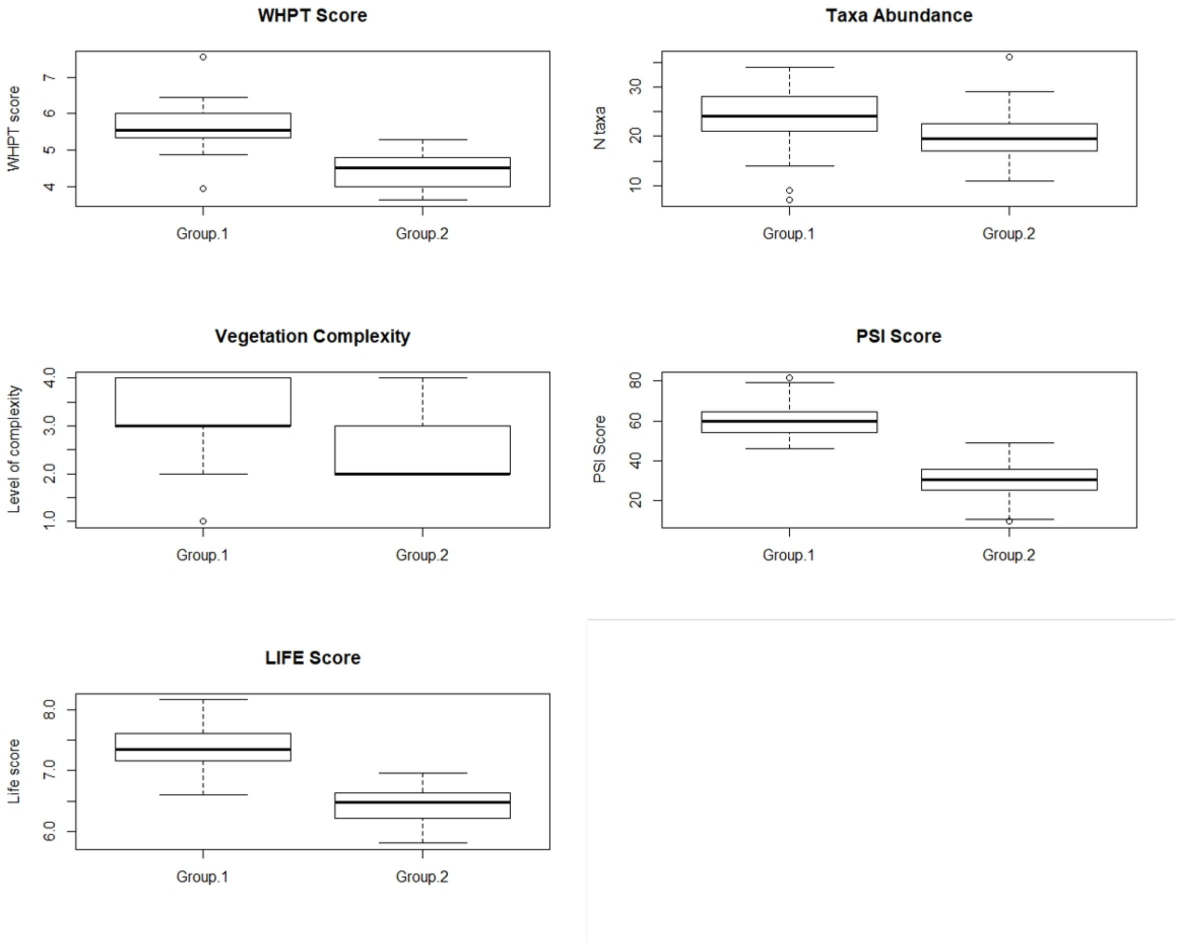


Figure. 5: Biological box plots clearly showing the difference between ecologically healthier group 1 and less healthy group 2, delineated using cluster analysis. Data is from EA monitoring data and field work. Group 1 N=20 and Group 2 N=29.

5.2.3. Chemical clusters

The chemical category produced two groups with an average silhouette width of 0.206. None of the continuous variables were found to be significantly different between groups (Table. 9.) but, group 2 contained all the sites with sewage treatment works whereas group 1 contained all the sites without sewage treatment works. Phosphorus was found to be marginally significantly higher in group 2 (Alpha = 0.1) but not to the extent that would be expected from sewage treatment works. Moreover, the median values for nitrogen, upstream roads and sewage treatment works, were higher for group 2 (Fig. 6.).

Table. 9: Comparison of the characterising variable means from chemical cluster analysis. The table makes it clear that neither group is significantly different with the exception of the presence or absence of sewage treatment works. Significance values are found from T-tests or Mann-Whitney U non-parametric tests after the normality assumptions, of T-tests, were checked with a QQplot and Shapiro-Wilk normality test. The alpha value selected was 0.05.

Measure	**Group 1**	**Group 2**	**Significance value**
N cases	32	10	
Phosphorus (mg/l)	0.10	0.21	0.096
Nitrogen (mg/l)	7.49	9.83	0.136
Dissolved Oxygen (mg/l)	11.00	10.46	0.101
pH	8.01	8.00	0.897
Water Temperature (^{o}c)	10.43	10.54	0.474
Road score index	31.16	37.80	0.436
Sewage treatment works**	0	1.30	<0.001*

* denotes significant at alpha 0.05

** denotes categorical variable therefore not a true mean average

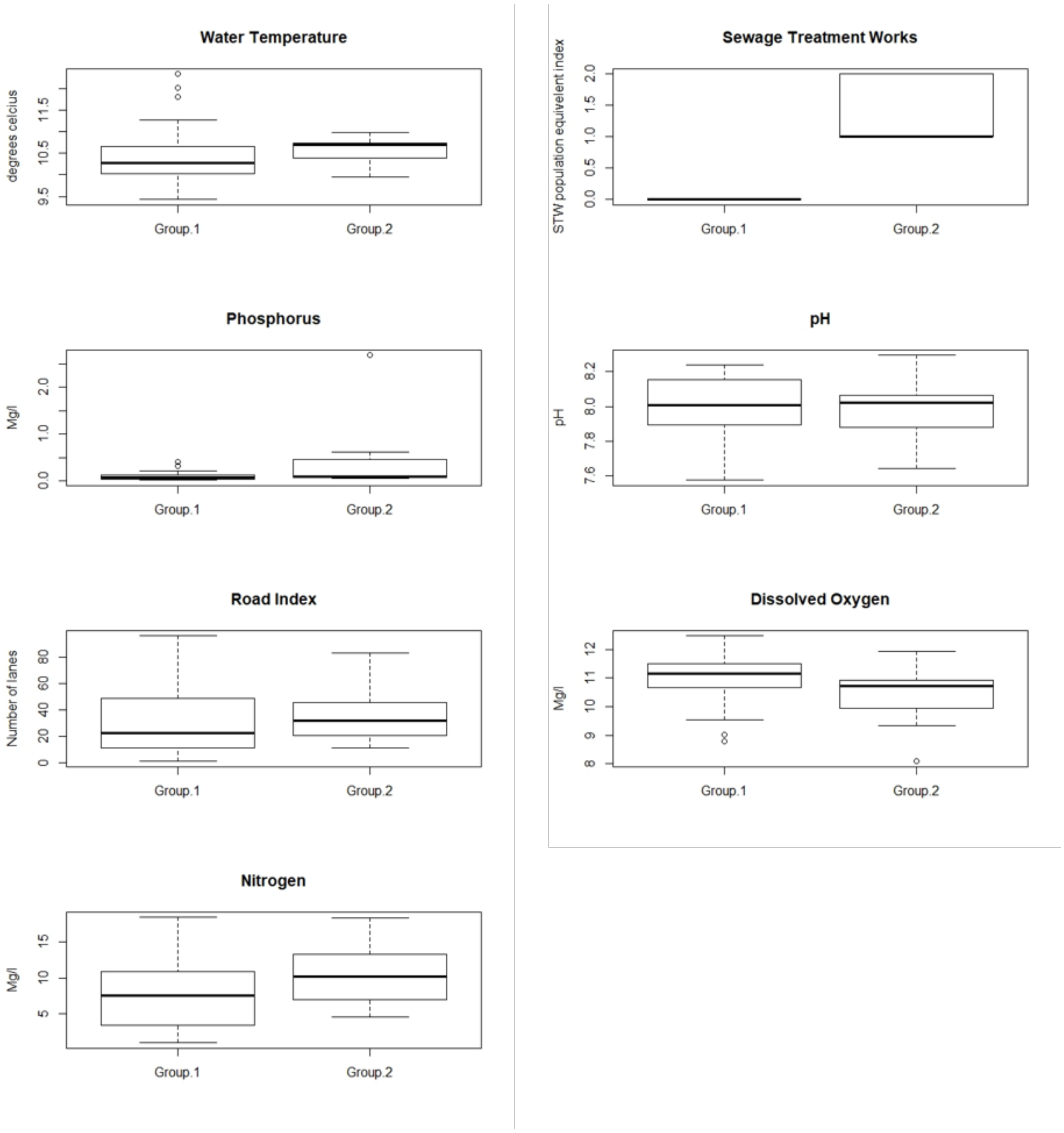


Figure. 6: Box plots displaying individual variable values which make up clusters created from the chemical category dendrogram. Sewage treatment works are a categorical variable hence the reduced variation around the median. None of these results are statistically significantly different (alpha = 0.05) suggesting high levels of variation clouding chemical trends. Group 2 contains all the sewage treatment works but these appear to have not impacted chemical levels. Group 1 N=32 and Group 2 N=10.

5.2.4. Hydrological clusters

Hydrological cluster analysis resulted in 3 clusters, with an average silhouette width of 0.234. These could be interpreted as low, average and high flow groups (Table. 10, Fig. 7.). Group 3 has high discharge values, high elevation and large catchment areas. Group 2 contains all the imperceptible flow occurrences and has less total flow types but has variable discharge (Q) values meaning that while it is more uniform, this is not necessarily caused by discharge reductions. Finally, group 1 has low base flow and Q50 but similar Q10 flows to group 2.

Table. 10: ANOVA and Kruskal Wallis tests showing that at least one of the 3 Hydrological groups is different in discharge levels. Along with flow characteristics, presence of flow and catchment area also shown to be significantly different at the alpha value of 0.05. Normality of data was tested, before statistical tests were chosen, this was done through a QQplot and Shapiro-Wilk normality test.

Measure	**Group 1**	**Group 2**	**Group 3**	**Significance values**
N cases	31	6	12	
Q10 (m3/s)	0.57	0.56	1.35	<0.001*
Q50 (m3/s)	0.17	0.22	0.43	<0.001*
Q95 (m3/s)	0.06	0.08	0.17	<0.001*
BFIHOST (m/Km)	0.62	0.64	0.51	0.103
DPSBAR	45.12	44.57	63.06	0.065
Catchment Area (Km2)	44.71	33.82	68.56	0.010*
Flow characteristics**	2.65	1.17	2.58	0.012*
Presence of flow**	1	0	1	<0.001*
Elevation (m above ordinance datum)	179.55	168.03	285.51	0.027*

* denotes significant at alpha 0.05

** denotes categorical variable therefore not a true mean average

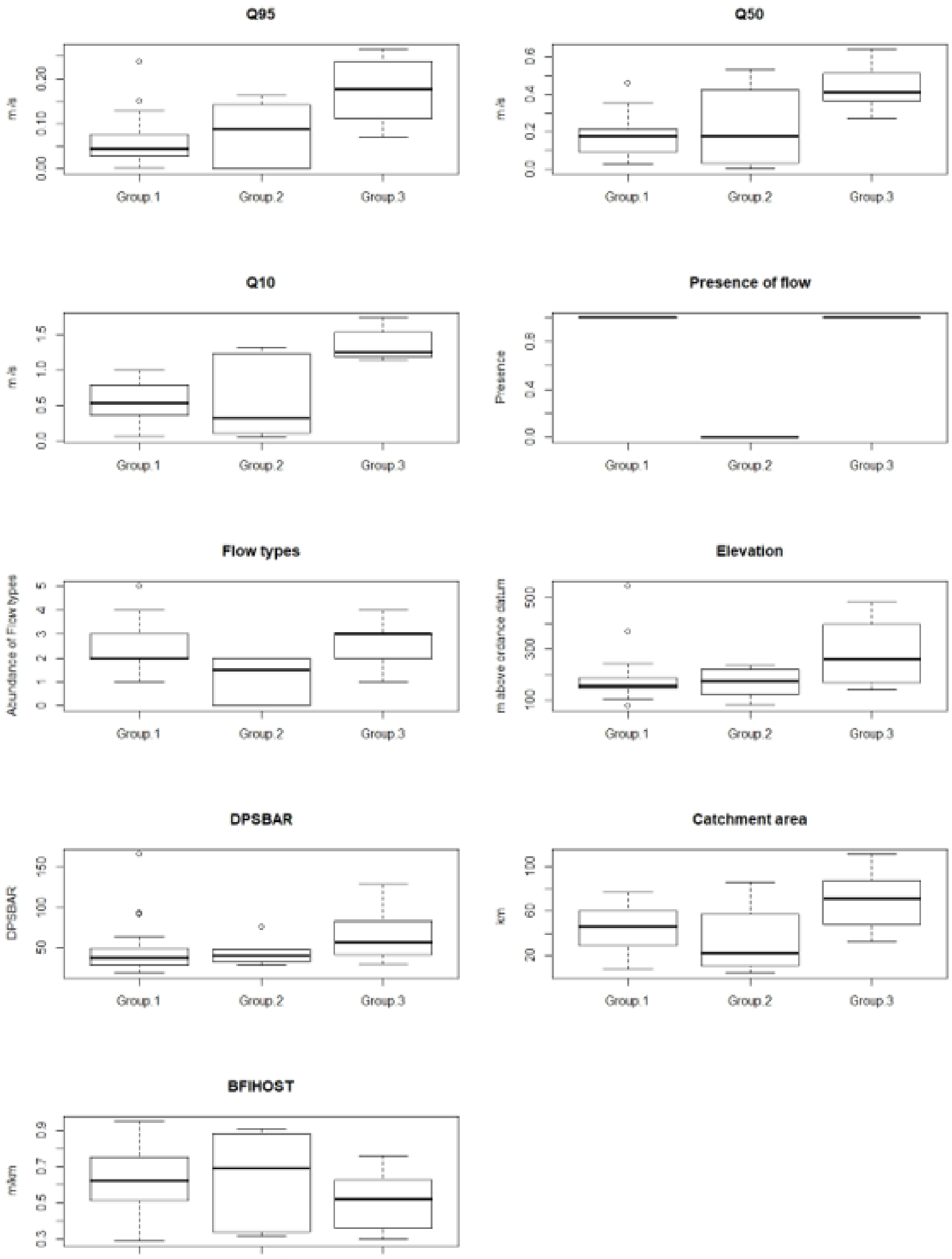


Figure. 7: Box plots showing the difference in hydrological cluster groups. Group 1 appears to be the hydrologically average river group, group 2 has all the low flow events and smaller catchment areas and group 3 has the higher catchment areas and higher flows. Data collected in the field and from the NRFA gauging system. Group 1 N=31, Group 2 N=6 and Group 3 N=12.

5.2.5. Physical clusters

The physical category was split into 2 clusters, with an average silhouette width of 0.210. Group 2 is the smaller group with only 9 cases which are heavily agricultural sites. This is supported by the sediment sizes being significantly lower for this group (Table. 11.). Land use variables do not show this clustering as clearly (Fig. 8.) but, this may be a result of the variation of land uses within the much larger group 1 (Fig. 9.). This variation appears to be constrained by other physical aspects of the system, which are more similar than land use.

Table. 11: Comparison of the characterising variable means from Physical cluster analysis. The table makes it clear that the smaller group, found to be mostly farms in the field, is only statistically different in sediment size and litter but not arable land uses in the catchment. Significance values are found from T-tests or Mann-Whitney U non-parametric tests after the normality assumptions, of T-tests, were checked with a QQplot and Shapiro-Wilk normality test. The alpha value selected was 0.05.

Measure	Group 1	Group 2	Significance value
N cases	41	9	
Sinuosity index	622.27	605.00	0.696
D50 (phi)	4.53	1.5	<0.001*
D84 (phi)	5.79	1.5	<0.001*
D95 (phi)	6.60	2.08	<0.001*
Width (m)	8.80	7.09	0.167
Depth (m)	1.55	1.56	0.968
Slope (cm/m)	0.01	<0.01	0.980
Visible bank erosion**	4.02	4.56	0.150
Anthropogenic litter pieces	3.56	4.67	0.011*
Habitat modification score	18.83	14.56	0.402
Urban land use (%)	12.54	5.93	0.411
Woodland land use (%)	10.19	9.46	0.772

Arable land use (%)	45.11	59.44	0.096
Grassland land use (%)	30.23	25.89	0.408

* denotes significant at alpha 0.05

** denotes categorical variable therefore not a true mean average

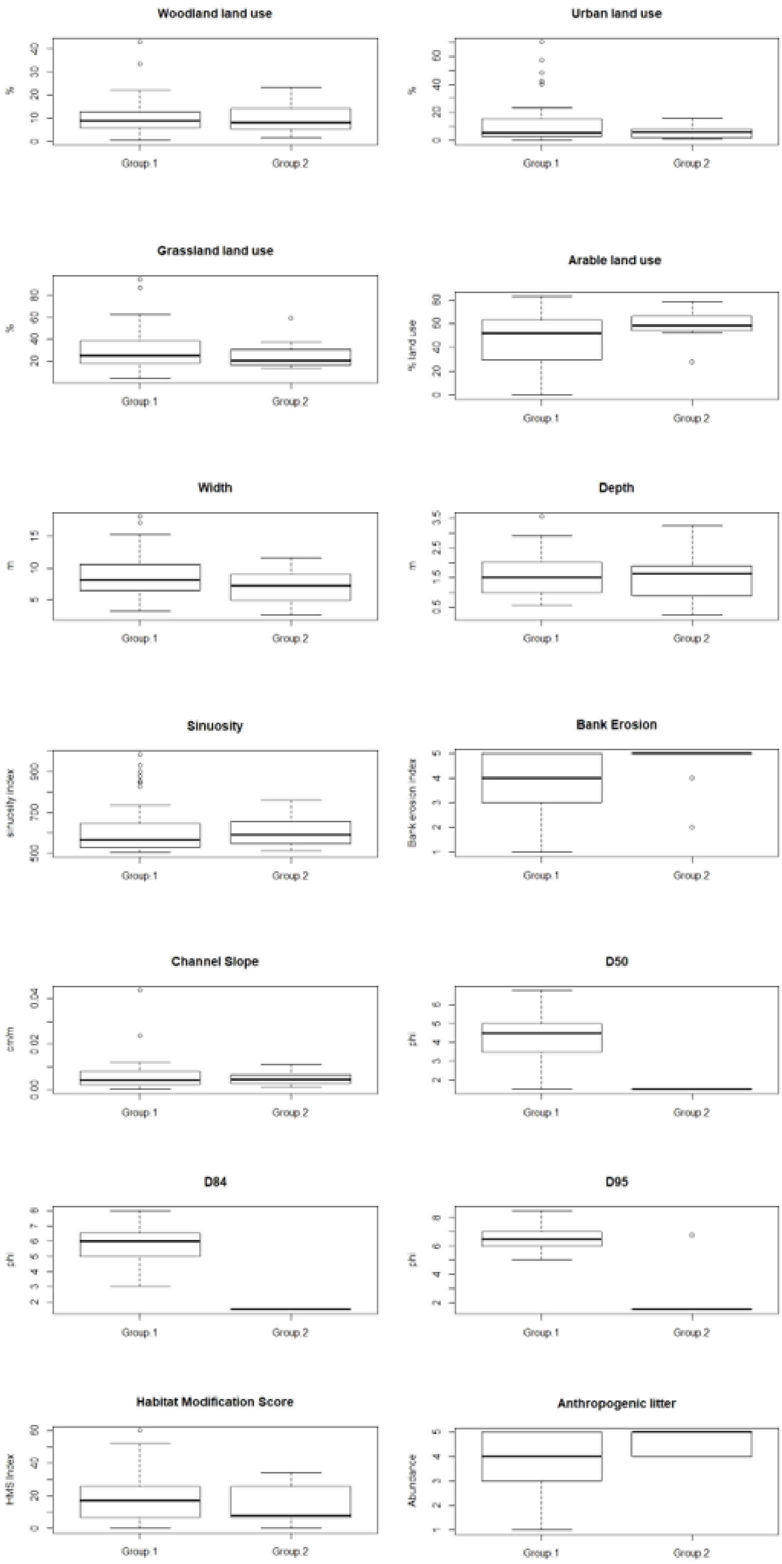


Figure. 8: Box plots of results from the physical category cluster analysis which has delineated an agricultural group (Group 2 N=9) and a larger more varied group (Group 1 N=41). The most evident difference is in bed sediment size which is entirely silt for the majority of group 2. It is interesting that an urban group was not also delineated by this physical cluster analysis.

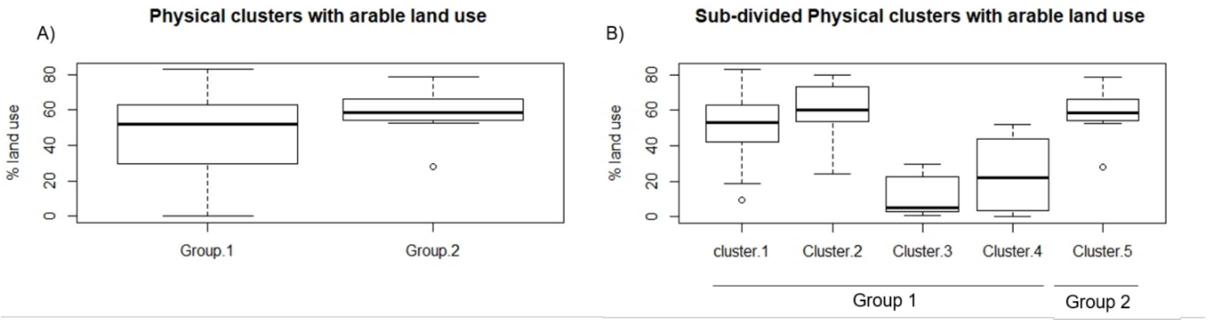


Figure. 9: Box plots isolating arable land use percentages for the clusters found from cluster analysis. A is the clusters chosen based on highest silhouette width, the medians do not look considerable different but the means of the two groups are different (alpha = 0.05, t = -2.2788, DF = 49, p = 0.03513). B uses 5 clusters to show that the groups of rivers within group 1 are very varied and therefore there are other rivers with similar farming extents to group 2 but they did not delineate as well due to variation in the variables most likely created from different variables.

5.4. River that are similar across different input variables

The clusters in each of the four categories contained different rivers. However, 27 rivers were found to co-occur in a cluster for each of the four categories of dendrogram (Table. 12.). These rivers were not all found together, but can be broken into four groups of rivers. Rivers in each of these four groups are therefore consistent in their biological, chemical, hydrological and physical parameters. This makes these four groups informative in the analysis.

Groups A and B are relatively similar, both representing good biological condition and average flow (Table. 13.) and are ultimately divided by the presence of sewage treatment works in Group 2. Group C is also similar to these first two groups, co-occuring in the biological and physical categories with good biological condition but, is different from group A and B in the hydrological grouping where it has higher flows. Therefore, hydrology still holds some characterising influence and these larger rivers are separated by this variable here. Finally, group D is made up of the most urban rivers in the study, although this has to be taken with some caution as none of the clusters directly predict urbanisation. It is interesting to note that group D representing heavily urban rivers, which were hypothesised to be delineated, represents only 3 rivers of the 50 (Table. 14.).

Table. 12: Co-occurrence groups. In all categories the groups of rivers displayed below co-occur in the same clusters. Table 13 displays which clusters each groups fall into. Some of these groupings are clear in their similarity such as Group D being heavily urbanised rivers. Whereas Group A for the most part is found in the larger clusters of each category.

Group A	Group B	Group C	Group D
Bailey Brook (1)	Blackfloss Beck (2)	Great Eau (11)	Bradford Beck (3)
Cringle Brook (5)	Dover Beck (6)	River Chater (28)	River Leen (34)
Dowles Brook (7)	North Brook(21)	River Hamps (32)	Spen Beck (47)
Hadley Brook (12)	River Gwash (31)	River Rea (38)	
Harpers Brook (13)	River Meden (36)	River Ryburn (39)	
Henmore Brook (15)	River Ryton (40)	Rothley Brook (45)	
Holywell Brook (16)	River Worfe (44)	Willow Brook (50)	
Lonco Brook (17)			
Mires Beck (20)			
River Dove (29)			

Table. 13: Cluster organisation of the co-occurrence groups. Numbers refer to groups on Dendrograms (Fig. 4.). It is clear that in some categories that multiple groups also co-occur together.

Cluster	Group A	Group B	Group C	Group D
Biological	2	2	2	1
Chemical	1	2	1	1
Hydrological	1	1	3	3
Physical	1	1	1	1

Table. 14: Co-occurrence groups with their ecological WFD classifications showing that there is no clear trend between the groups. Heavily Modified Water Body status is also indicated with the only real pattern being group D, the heavily urban river type, being entirely classified as HMWBs.

A	B	C	D
Moderate	Poor	Poor – HMWB	Moderate – HMWB
Good	Moderate	Poor	Moderate – HMWB
Moderate	Good	Moderate	Moderate - HMWB

Poor	Moderate	Moderate - HMWB
Moderate	Moderate	Moderate
Moderate - HMWB	Moderate	Moderate
Moderate	Poor	Moderate
Moderate		
Moderate - HMWB		
Moderate		

5.5. Extent of categorical alignment

Few rivers were consistently found together across different categories of cluster analysis (Table. 15.). Only 11.3% of pairs co-occurred in different categories, suggesting large variability between either the rivers or that one class of parameters is not able to explain variance in another. The complete table of category similarities is presented in the Appendix (D.). The hydrological category appears the worst category for predicting river type shown by other categories as it correlates the least with the other categories but, this may be a statistical artefact of hydrology having 3 clusters as opposed to 2 clusters in each of the other categories. The increase in groups reduces the likeliness of river combinations occurring by chance in the same group. The opposite may be true in the case of the physical category having a single larger group (n=41) meaning rivers are more likely to be in the same group regardless of similarities in other categories.

Table. 15: Table displaying the percentage of similarity between different groups of clusters and overall similarity. To some extent this is influenced by larger clusters in some categories than others.

	Biological	**Chemical**	**Hydrological**	**Physical**	**Total**
Biological	N/A				11.30
Chemical	31.45	N/A			
Hydrological	27.00	25.80	N/A		
Physical	38.86	41.75	34.69	N/A	

To further investigate alignment between the clusters between different categories, the dissimilarity between sites was calculated for each class of variable using the Gower dissimilarity index. Dissimilarity scores for variable classes were plotted against one another and correlation coefficients calcultated (Fig. 10.). These dissimilarity scores measure the difference between every catchment in the study and the correlations measure the difference between the dissimilarity scores of each category. Therefore, if the clustering in two different categories was identical, the points would lie on a 1:1 line. Significant correlations exist between the biological category and other categories but, with only poor correlations (r < 0.202). The only other significant result was between the hydrological and physical categories, which despite having the strongest correlation (r = 0.3), still represented only a weak relationship.

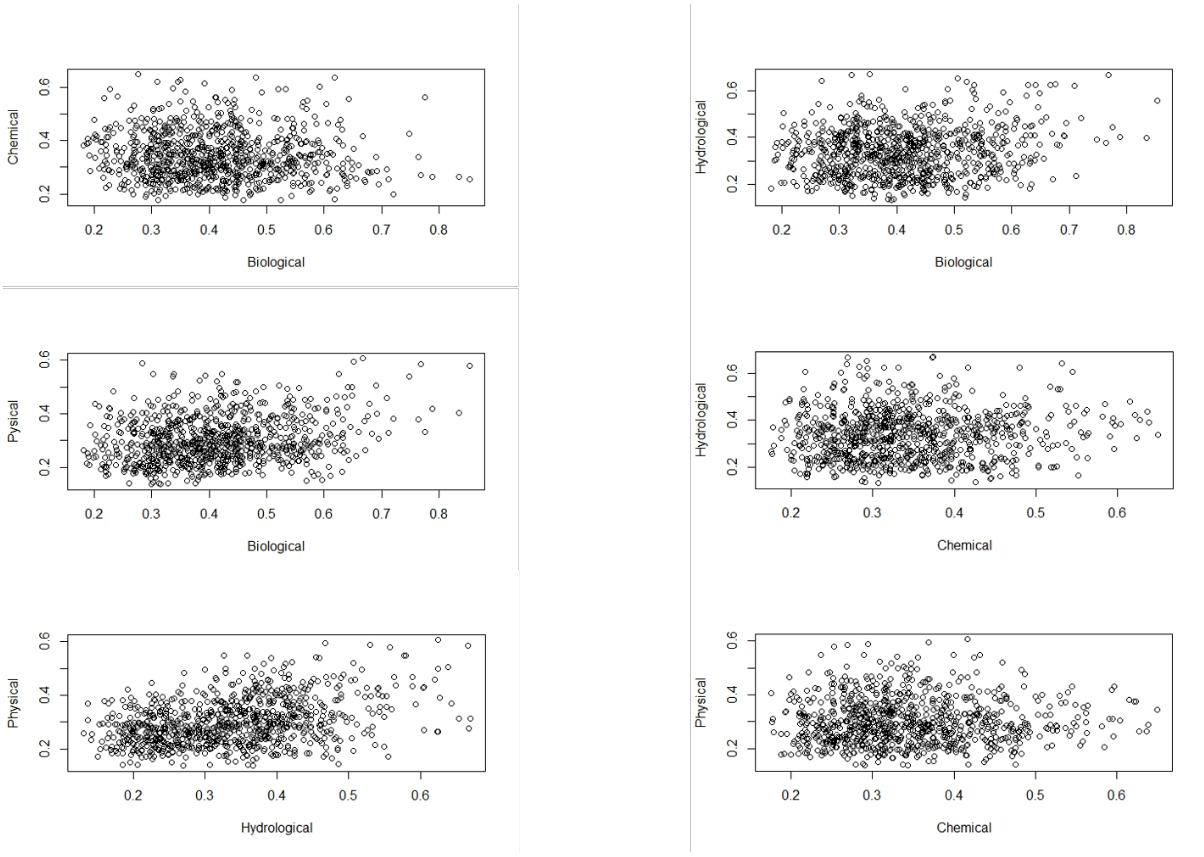


Figure. 10: Scatter graphs showing the dissimilarity between different river pairs in different clustering categories. All dissimilarity measurers were created using the Gower dissimilarity matrix with the Vegan package on R. full statistical results are shown in table. 16. However, BP, HP and BH are all statistically significantly correlated with an alpha value of below 0.05.

Table. 16: Correlation and significance results for Gower dissimilarity comparisons between pairs of streams in different variables. It is clear that there are no clear positive or negative correlations between categories with R2 values never surpassing +/- 0.4. This suggests that there is no strong correlation between different categories. Rivers may be more varied than in the past and other drivers may be present which were not before.

Matrix	Pearson's r	Significance
Biological – Chemical	-0.058	0.092
Biological - Hydrological	0.106	0.002*
Biological - Physical	0.202	>0.001*
Chemical - Hydrological	0.059	0.085
Chemical - Physical	0.023	0.495
Hydrological – Physical	0.333	>0.001*

5.6. Clusters correlations to the Water Framework Directive

Mapping the WFD biological score for each site onto the clusters produced for each of the four categories led to no discernible pattern (e.g. Fig. 11.), suggesting there is a disconnect between WFD measurements and the streams they are classifying, at least for those sites studied here. However, the most remarkable finding, when considering WFD assessments, is the complete lack of correlation between the biological clusters which have been created and the ecological WFD classifications (Fig. 11.). This is because there is a clear higher and lower biological group therefore it would be expected that there would be some alignment with the lower WFD classifications.

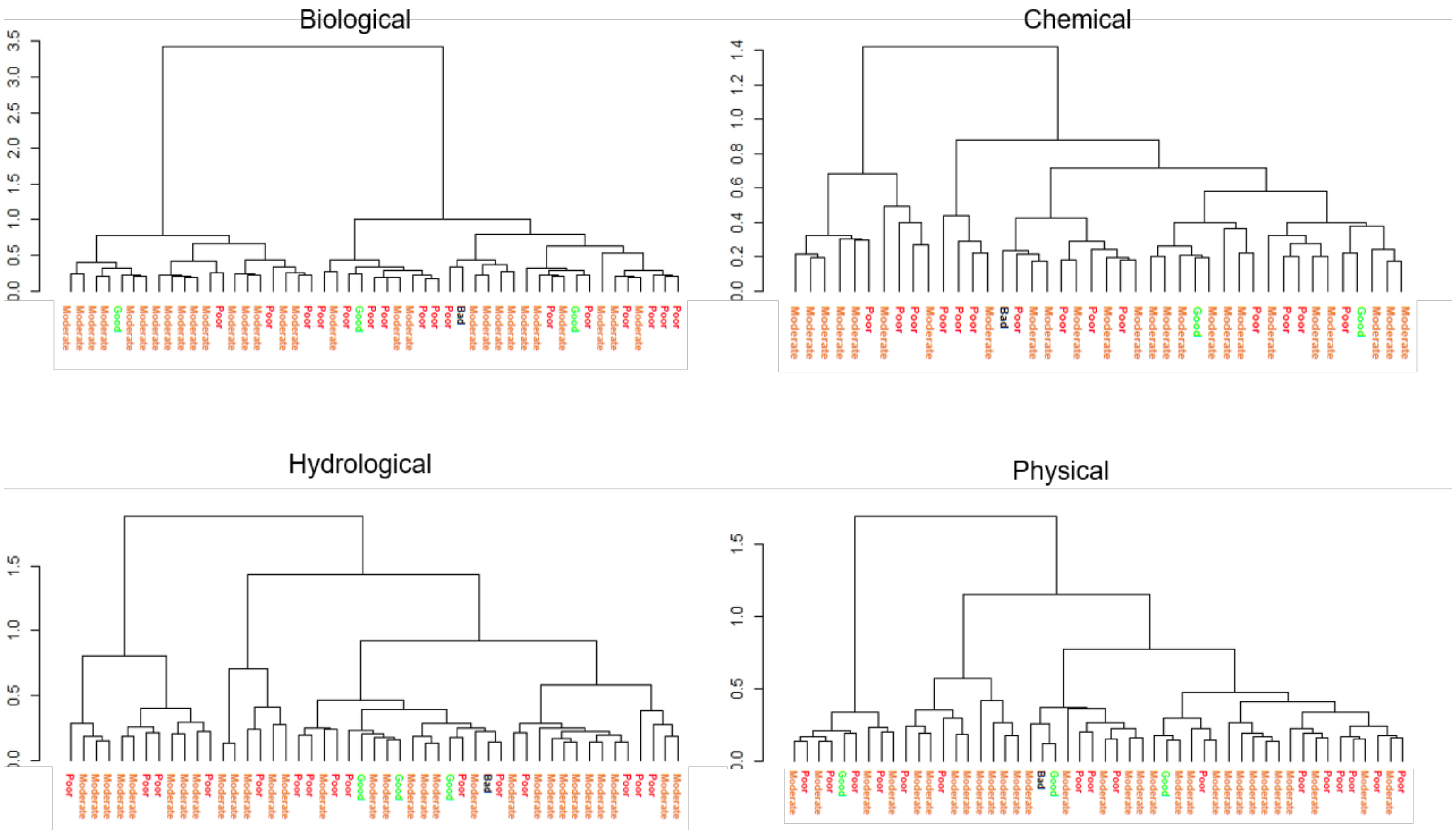


Figure. 11: European Union Water Framework Directive ecological river scores sorted by categorical cluster analysis. The same figures were not created for WFD chemical scores as 49/50 were considered of a "high" standard. It is clear from these plots that there are no strong patterns between the results of categorical cluster analysis and WFD scores. It is also worth understanding that all these streams were supposed to reach a "good" status by 2015 (McGinnity, 2002) and there are only 3 out of 49 that have done so. Moreover, it would be expected that the biological category would show a clear trend with WFD scores due to a statistically significant good and bad cluster but this trend has not occurred.

6. Discussion

6.1. Overview of findings

The use of cluster analysis to visualise the dissimilarity of streams in the Midlands has marked some groups of streams as substantially different from the full sample to warrant their own stream types. Some of these stream types, such as heavily urban streams, are found consistently regardless of category of analysis. Other stream types are only applicable when looking at the rivers through the lens of one individual category.

The variability in clustering between categories of data is important because it implies shortcomings in current management techniques. The assumption that managing for one goal will positively affect the rest of the river system may not be the case in heavily impacted rivers. The disconnect between the physical, hydrological, chemical and ecological condition of these streams may also be a symptom of the novel ecosystems that exist in these altered rivers and implies new science may be needed to effectively manage these systems (Hobbs et al., 2009). The findings also demonstrate that current classification systems are not appropriate for rivers and catchment conditions in the Anthropocene. Driving variables and key controls on river form and process have only limited impact on the typology developed here, indicative of river processes that are disjoined from the expected natural relationships (Downs and Gregory, 2014). Arguably, some driving variables have been limited in their power but, still have some influence on river type in the context of anthropogenic management.

6.2. Relationships between descriptive variables

The correlation matrix (Fig. 3.) was investigated for its adherence to the literature (sections 3. and 4.). Any deviations from the literature will have implications on how cluster analysis findings are interpreted. In order to correctly analyse these findings it has to be noted that land use variables are taken as percentages of the catchment and, therefore, are mutually exclusive to other land uses in the same catchment. This may have diluted the results as there is now the possibility for auto-correlation. For example, the lack of urban land use may be driving the correlation seen in a case that has an abundance of grassland land use, thus leading the researcher to incorrectly assume that it is the abundance of grassland causing the correlation found.

Arable land use was found to have positive correlations to nitrogen levels, BFIHOST and temperature. This fits with the literature, with concerns over the effect of fertiliser and annual manure on elevating nitrogen concentrations (Dodds and Smith, 2016) with implications for in-stream ecology (Jarvie et al., 2018). Temperature is likely related to the lack of riparian vegetation where economically productive land has been maximised, minimising the shade cast by marginal trees (Johnson and Wilby, 2015). BFIHOST suggests surface runoff is also high in these areas, potentially due to soil compaction and agricultural drainage to increase land productivity. Negative correlations also exist between arable land to dissolved oxygen, WHPT, PSI, LIFE score and sediment size making it clear that a lower environmental quality would be expected as a result of the types of practises associated with arable land in the study area.

Clear positive correlations occurred between urban land use, discharge values and river widths, despite there being fewer urban rivers in the study. This supports the concept of the "hosepipe effect" in which urban rivers have been turned into drains to remove water from cities as quickly as possible (Navratil et al., 2013). They have often been artificially over-widened to convey a greater proportion of the hydrograph (Downs and Gregory, 2014) and therefore, it was expected that clusters of urban rivers would form.

Finally, grassland showed a positive correlation with elevation, discharge values, PSI, LIFE, and WHPT scores. Grassland also showed a negative correlation to catchment area, temperature and nitrogen levels. There are three factors at play here. First, grassland includes sheep grazing which predominantly occurs at higher elevations where arable farming is not economically viable. Hence, these are elevated first order streams with flashy discharge from a small catchment area (Strahler, 1957). Secondly, the negative impacts of grassland are relatively limited and, by being elevated first order streams, there is little opportunity for detrimental chemicals and other pollutants to accumulate and damage ecology (Wohl, 2017).

Most surprising of all the findings in this subsection, was that the macroinvertebrate indices do not necessarily correlate to the values that they were designed to represent. LIFE scores do not correlate to the discharge scores, which may relate to the scale of influence with LIFE working at a patch scale (Frissell et al., 1986); (Extence et al., 1999). PSI also does not correlate to sediment size, perhaps because bed sediment size was sampled whereas PSI responds more to fine sediment in the water column and deposited on the surface. Dissolved oxygen does not strongly correlate to any other variables despite having been shown to be a strong one value alternative of the water quality index (Rudolf et al., 2002) and may correlate to this as much as 90% (Kannel et al., 2007). This would suggest there are either some issues in the measurement of dissolved oxygen or that none of the other variables impact water quality. The WFD is heavily reliant on these measures for river quality classification and, therefore, the results here question the basis of these scores. While this study does not directly attempt to validate these measures, these results do question whether other factors are also influencing invertebrate community indices.

6.3. How clear were the clusters and river types found?

There was a relatively low number of clusters in each category, suggesting the methods used were strong enough to delineate between different types of river but not precise enough to pick up more subtle trends. The more conservative river types used here, which had the clearest differences, are potentially the most widely applicable and more subtle differences are unlikely to yield management benefits. To potentially increase the number of clusters, additional sites would ideally be incorporated into the analysis.

Most of the categories produced one large and one or two significantly smaller cluster(s). This suggests that the majority of the rivers are relatively similar or at least, the within cluster variation is too great to define any group of sites as different from the others. However, in each case the smaller group is informative as a relatively constrained group of similar sites. This shows that in each category there are river types that are significant enough to divide a set of rivers from the total sample.

6.3.2. Biological clusters found

Biological clusters were more evenly sized, and there appears to be better and worse group for ecological health (Table. 8.). From this, it could be inferred that many rivers are struggling ecologically, which is well known (Reyjol et al., 2014, Jarvie et al., 2018). The fact that the biological river types do not necessarily align with other categories will be discussed further but, it is noted here there are multiple different situations in which a river type can be considered comparatively ecologically healthy.

LIFE scores being higher in the second group suggesting that it has more variable and higher flows at the patch scale (Frissell et al., 1986). One of the causes of this is anthropogenic factors which can impact the flow regime (Laini et al., 2018) and shift riverine ecology (Arnell, 1996). Any correlations with the physical category would be important in this regard as this is most commonly associated with impoundments (Krajenbrink et al., 2019). Thus, an unhealthy ecological group may be a clear indicator of anthropogenic change to stream biology.

6.3.3. Chemical clusters found

Chemical clusters appear to be defined by the presence or absence of sewage treatment works but, not necessarily the measured chemical concentration of the rivers. It would have been expected that a grouping streams with STWs present would be degraded chemically (Civan et al., 2018). None of the considered chemical differences were significant (Table. 9.). This is likely to be an artefact of data collection producing very variable chemical values due to non-standardised time of sampling and low temporal resolution (EA, 2019). It is possible that higher resolution data, or parameterisation that can separately quantify chronic and spiked influences of chemical pollutants, would provide a better basis for typologies. However, such measurements are rare, which is one of the main reasons why biomonitoring scores are used to assess water and habitat quality.

It is already well known that the majority of streams in the UK are degraded by anthropogenic influence (Everard and Powell, 2002) and over-abundance of Phosphorus is an issue in small streams (Jarvie et al., 2018). Being driven by urbanisation, STW and intensive agriculture (Bowes et al., 2018)further support the analysis that either the data is not sufficient or all streams in the study are equally heavily degraded despite WFD classifications of High chemical status (49/50 streams in the study).

6.3.4. Hydrological clusters found

Hydrological clusters are more distinct than for the other categories. This suggests that hydrology plays an important role in driving river types regardless, even where the input rivers are heavily impacted by anthropogenic measures. In the same vein, hydrology being driven by anthropogenic measures supports the concept of anthromes (Acreman et al., 2014). However, environmental flows and anthropogenically influenced flows are never likely to be as variables as the natural regime (Tharme, 2003). Therefore, what was needed to be tested was variability in the hydrograph, which is not possible for a study of this size. Moreover, should the conclusions of this study be true and there is no natural analogue then it would be impossible to test if the hydrology of these streams had become more homogenous.

In the larger group the presence of anthropogenic activity may be visible in the discharge statistics (Fig. 7.). In most cases the average river group (Group 1) has a higher Q10 median than Group 2 (the low flow group) but has similar or lower Q50 and Q95. It is hypothesised that this is the result of water abstraction which would not be noticeable at high flows. Moreover, if the majority of group 1 rivers have higher flood flows it would be expected that their usual flows would be higher than group 2 without an outside influence.

Elevation being higher for the high flow group 3 would be unexpected as high elevation is associated with smaller catchments and therefore lower flows (Charlton, 2007). At the scale of this study elevation cannot be anthropogenically impacted therefore if hydrology is dictated unexpected by elevation it would suggest that the trends are not the result of anthropogenic influence. Nevertheless, the statistic used was highest elevation in the catchment and so it appears to be a proxy of the larger rivers in the study and therefore does not detract from the conclusions made.

6.3.5. Physical clusters found

It was hypothesised that rural farming streams and urban streams would be delineated in the physical category; however, this was only partly the case and it proved that sediment size was the main delineating factor. Nine farming streams were separated from the other 41 streams (Appendix. E.) which suggests that urban streams, at least in terms of width and depth are not different enough to warrant a separate category. It is also worth noting that many of the rural streams were also widened with straightened channels with re-shaped banks, reducing differences in channel morphology between stream types.

There were four groups of rivers that remained in the same group regardless of category of variables used to define grouping (Table. 13.). Three of these were relatively similar. Because of the large variance between sites within large groups of rivers, it is possible some rivers can be in the same group but be relatively dissimilar. These four groups were found in the larger categorical clusters, which means pairs of rivers are more likely to be found together through chance alone. However, the fourth group (group D) is a smaller group of 3 rivers, which are the most urbanised rivers in the study and remain consistently separate from most other rivers. These streams are found to be in the lower biological group, which fits with the literature for urbanised streams (Fenn, 2018).

6.4. Comparing Classifications

Based on the cluster analysis we found that site groupings were largely based on: whether a river is heavily urbanised, whether it is in an area of arable farming; their relative discharge; their biological condition; and the presence of sewage treatment works. Despite the importance of these parameters, there was considerable variation between rivers making it hard to delineate distinct river types. However, the worth of the proposed river types classified here is demonstrated when compared to previous classification systems.

Cochero et al. (2016) used a priori decision making that helped split urban and rural into two separate groups. To some extent, our results support this conclusion although there was much more variation between rivers, suggesting a continuum of modification is more appropriate than a firm threshold. Therefore, classification systems that use a clear divide between urban and non-urban streams, at least in the UK, are potentially missing a lot of information and variability in stream type. In addition, rural streams were also heavily impacted by anthropogenic influences and, rather than considering urban streams as being “bad” (Davenport et al., 2004), instead considering the range of human impacts across anthromes and using typologies to direct possible management options.

Hydrology still appears to be a defining driver in river systems as it has consistently been predicted to be (Knighton, 2014). However, the casual relationships between the physical attributes of a river channel and its hydrology, as documented in seminal papers (Leopold and Wolman, 1957), is no longer the case in the Anthropocene. Hydrology may still be able to define the broad characteristics of a river but, the vast majority of the rivers in this study have been modified and manipulated to the extent that the dominant controls on bed sediment, channel width, sinuosity and depth are is not hydrology but human activity (Goudie, 2016). As a result, hydrological context remains integral to river classification and should be included in classification systems but, it may now not be as significant as for wild rivers.

As well as the hydrological aspects of bed sediment being anthropogenically driven, anthropogenic litter at study sites was ubiquitous, often changing the D84 and D95 which would otherwise have been totally comprised silt. It is also noted that rural streams probably have such high levels of silt sediment as a result of human land use change, meaning even the mineral sediment component of these systems is not truly natural (Walling, 2005, Taylor et al., 2000).

One area where this study agrees with current classification systems is the importance of catchment scale land use. The whole catchment is important to the river system (Brierley and Fryirs, 2013) but, due to co-variance its inclusion does not add significantly more explanatory power, which is partly why it was not included in RS (Brierley and Fryirs, 2000). Firstly, the 9 streams delineated by physical conditions are agricultural streams, with significantly higher arable land extent than other streams (alpha = 0.05, t = -2.2788, DF = 49, p = 0.03513). However, when looking at box plots the dominance of agriculture, in group 2, is less clear (Fig. 9.). The number of clusters selected can hide a lot of variation in some variables. Had five clusters been deemed the most appropriate then differences in arable land % between clusters would be clearer (Fig. 9.).

Other rivers having high agriculture land use values (Fig. 9.) but not being in the agricultural river group is telling. It corresponds with the assessment that land use has greater influence closer to the stream (Cochero et al., 2016) as can be seen in images of the sites (Appendix. E.). This may be because anthropogenic factors, such as roads that alter sediment and water run-off, disrupt signals from catchment scale land use. Moreover, the impacts of farming are modulated by other variables, for example, more sediment is likely to be inputted from arable land when there is also a lack of riparian vegetation. The disruption of clear trends and interaction of variables highlights a draw back in the method.

The River Styles framework identifies valley confinement as the main delineating factor between different types of river. There was only one confined valley in this study - the River Dove yet, this river was not placed in a distinct grouping from other sites. This may have been because the fieldwork location was at the downstream end of the valley confinement and because the site includes a ford and footbridge and was, therefore, anthropogenically influenced despite its otherwise natural looking aesthetic. However, it appears that biology and hydrology may still be more dominant controls of the river system in this case (Castro and Thorne, 2019). It could be argued that the variety of anthropogenic influence across the dataset has removed valley confinement as a significant determinate in this study. In addition, straightening rivers and disconnection from floodplains has given many anthropogenic streams similar characteristics to confined streams (Hammer, 1972). While it is accepted that river styles would not be directly applicable in the UK, having been designed in the significantly different geology and bio-climate of Australia, the main reason that there is no alignment over valley confinement is that of anthropogenic induced variation across the whole study.

6.5. Differences between categories of environmental variables

As discussed in section 5.5, there is only 11.3% similarity between the groupings across the four categories, which is considerably more variation than originally hypothesised. This conclusion is further amplified by the lack of Pearson's correlation between different categories.

For the chemical category, patterns could not be distinguished from random, implying that the measures used to create the chemical category were the weakest of the study. This finding is important as the WFD relies on chemical and biological data to assess the damage currently occurring in UK rivers, and assess management options (Brack et al., 2017). This is especially apparent as 49 of the 50 rivers are shown to have high chemical status (Malaj et al., 2014) but, overall only 3 achieve good status (none achieve high status) because of detrimental ecological condition.

Hydrology and physical categories were correlated and provide intuitive groupings, suggesting that they still exert a form of control on river type, even when rivers are heavily impacted by people. Anthrocentric catchments are expected to have rivers designed based on their social and economic context, rather than hydro-geomorphic processes. Where flood damage is likely to be high, channels are designed to transfer water quickly and to convey high discharges. Therefore, whereas links between hydrology and channel form are direct in natural streams, they are indirect in the Anthropocene as humans design channels to convey a greater range of flows. Similarly, dams and weirs increase base flow by pooling water (Poeppl et al., 2015), also linking hydrology to anthropogenic physical attributes.

The physical category also shows a significant and slightly positive correlation to biological category (Table. 16.), suggesting that to some extent there is still a relationship between physical habitat and organism diversity. However, the weakness of this correlation supports the many arguments against the assumed significance of physical habitat creation without supporting considering of ecological functioning (Palmer et al., 2010). Remnants of brick and concrete structures, such as old dams, create heterogeneous habitat in otherwise homogenous urban and agricultural streams. As such, novel ecosystems may be present and contributing to biological-physical interactions in unexpected ways (Hobbs et al., 2009).

6.6. Management implications of anthropogenic streams

The lack of strong correlation between categories and the different river types identified, potentially have considerable ramifications for river management. Many assessment methods exist to judge the state of a river and whether it should be considered environmentally or socially sound, depending on the goals and context of that stream (Fernández et al., 2011, Carvalho et al., 2011). However, these methods rarely contextualise which areas are consistently failing standards that have been set by the WFD or other management protocols, and where attention should be prioritised. Finding different clusters of rivers when analysed with different categories infers that many restoration and management projects will not have the results intended. Goals designed around one category are expected to prioritise that category but also improve the whole system. Our findings suggest that this is not the case and goals focusing on one category may negatively affect other categories

The WFD classifications mainly rely on biological considerations making the assumption that this is the end point of other categories, as they all have some significance biologically (Kallis and Butler, 2001). However, as a comparison between groupings show this may not directly be the case and the amount of variation in these categories could be hampering the ability of the EA to meet WFD targets (Hering et al., 2010). It may also suggest that the threshold for other factors such as chemistry is currently set too leniently (Brack et al., 2017). The biological category used here also does not align well with the ecological assessment of the WFD (Fig. 12.). As clear healthy and unhealthy clusters were identified here, it could that considerable levels of degradation are hidden from WFD assessments. Moreover, riparian is the primary provider of nutrients into the water column, through litter (Vannote et al., 1980). This forms the basis for aquatic life but is not incorporated in WFD assessments. Therefore, the lack of alignment between the biological clusters and WFD ecological classifications suggests integral issues with WFD measurement and ecological thresholds.

WFD classifications do not reveal any patterns when presented alongside the groups of co-occurring rivers (Table. 15.). It is interesting that the heavily urbanised group has moderate ecology based on the EA classifications, despite being found in the most modified group of rivers. Although using 50 sites was sufficient for statistical determination of groupings with the chosen variables, to further explore observed trends, a wider range of streams, ideally from across the England, would be tested. This would also enable differences in regional land use and management to be studied and to enable the incorporation of invasive species impacts and susceptible. Furthermore, it would provide better opportunity to test the novel ecosystem hypothesis and where biodiversity is made stronger by the inclusion of non-pest invasive species.

7. Conclusion

The project aimed to create a river typology accounting for anthropogenic impacts and assess if the drivers of this typology were still morphological in origin. There was some success in creating a typology, although it could not be considered comprehensive due to the limiting effect of substantial variability within the dataset. The key findings were that clustering did produce clear cases (objective 1), and for the first time, different clusters of the same sites were found depending on the focus of the input variables (objective 3). It is hypothesised that this is because anthropogenic alterations disrupt the linkages between the physical, hydrological, chemical and biological spheres, and that typologies developed on wild, unaltered rivers would be more similar between these four input categories. This result has management implications because the assumption that improving hydrology or physical channel form will have positive impacts on other facets of the river environment might be misplaced. Instead, the results here suggest managing a specific single category is unlikely to lead to improvements in river environments and that river restoration needs to simultaneously consider all four categories.

Another implication of the disconnect between categories of cluster is that some past classification systems are unlikely to be appropriate to anthropogenic rivers (objective 2). For example, Rosgen relies on physical metrics to determine river type, with the implication being other factors are driven by physical form (Rosgen, 1994). There was some evidence of a link between hydrology and physical clustering but, these are more likely indirect as a result of human interference in river form, rather than the direct linkages between hydrology and channel form defined in classic work on river processes (Leopold and Maddock, 1953, Montgomery and Buffington, 1998).

Anthropogenic drivers of these clusters were varied (objective 4) although some key trends were evident. Arable farmed rural streams were often grouped together, although there was great variation within this group due to other anthropogenic pressures that co-occur, such as sewage treatment works. There was also evidence of distinct urban and rural stream clusters but, these were found to be weaker than previously documented and expected. While dividing rivers in to urban and rural may be useful in some contexts, there is considerable variation within the two groups and they are perhaps better represented by a continuous scale of anthropogenic influence. In response, it is suggested that the terms: urban and rural, should refer to socio-economic constraints on future management rather than depicters of current river type.

These results offer a proof-of-concept that demonstrates that the controls on river form and process are changing and that traditional river typologies dominated by natural fluvial processes are unlikely to be representative of many rivers today. To improve the classification, a greater number and range of the study sites is needed. These may strengthen existing groups and also increase the opportunity for further sub-division of groupings. Particularly important groups not currently included would be intermittent streams in the South-east of England, which are heavily affected by water abstraction (Westwood et al., 2017). In addition, including a greater range of river sizes would add to the analysis, and allow an assessment of whether clusters are scale dependent. Finally, repeating the analysis through time to test the temporal strength of clusters would provide an important sense of cluster strength (i.e. do clusters change year-to-year) and also, allow identification of whether clusters are systematically changing through time due to continued land-use and climate change.

The results suggest an urgent need to understand the controls and drivers in anthropogenic streams, and a reassessment of management aims. Whilst this study supports past work that human impacts are detrimental to river environments, here a gradation of change from rural to urban streams and, ecological and chemical condition does not obviously relate to WFD thresholds. The disconnect between physical, hydrological, ecological and chemical data indicates that rivers in the Anthropocene are facing multiple pressures but do not necessarily respond to all of them at the same scale, time or speed and that expected linkages between these categories might be allusive, with implications for how to focus management and holistic approaches to restoration.

8.0. Reference list

ACREMAN, M., ARTHINGTON, A. H., COLLOFF, M. J., COUCH, C., CROSSMAN, N. D., DYER, F., OVERTON, I., POLLINO, C. A., STEWARDSON, M. J. and YOUNG, W. (2014). Environmental flows for natural, hybrid, and novel riverine ecosystems in a changing world. *Frontiers in Ecology and the Environment,* 12**,** 466-473.

AGASSI, M., MORIN, J. and SHAINBERG, I. (1990). Slope, aspect, and phosphogypsum effects on runoff and erosion. *Soil Science Society of America Journal,* 54**,** 1102-1106.

ALLAN, I. J., VRANA, B., GREENWOOD, R., MILLS, G. A., ROIG, B. and GONZALEZ, C. (2006). A “toolbox” for biological and chemical monitoring requirements for the European Union's Water Framework Directive. *Talanta,* 69**,** 302-322.

ANDERSON, R. J., BLEDSOE, B. P. and HESSION, W. C. (2004). WIDTH OF STREAMS AND RIVERS IN RESPONSE TO VEGETATION, BANK MATERIAL, AND OTHER FACTORS 1. *JAWRA Journal of the American Water Resources Association,* 40**,** 1159-1172.

ARCEMENT, G. J. and SCHNEIDER, V. R. (1989). Guide for selecting Manning's roughness coefficients for natural channels and flood plains. US Government Printing Office Washington, DC.

ARMITAGE, P., MOSS, D., WRIGHT, J. and FURSE, M. (1983). The performance of a new biological water quality score system based on macroinvertebrates over a wide range of unpolluted running-water sites. *Water research,* 17**,** 333-347.

ARNELL, N. W. (1996). *Global warming, river flows and water resources*, John Wiley and Sons Ltd.

ATTARI, M. and HOSSEINI, S. M. (2019). A Simple Innovative Method for Calibration of Manning's Roughness Coefficient in Rivers using a Similarity Concept. *Journal of Hydrology*.

BARRON, O. and BARR, A. (2009). Effect of urban development on water balance in the Southern River catchment.

BATHURST, J. C., HEY, R. D. and THORNE, C. R. (1979). Secondary flow and shear stress at river bends. *Journal of the Hydraulics Division,* 105**,** 1277-1295.

BEISEL, J.-N., USSEGLIO-POLATERA, P. and MORETEAU, J.-C. (2000). The spatial heterogeneity of a river bottom: a key factor determining macroinvertebrate communities. *Assessing the ecological integrity of running waters.* Springer.

BENDA, L., POFF, N. L., MILLER, D., DUNNE, T., REEVES, G., PESS, G. and POLLOCK, M. (2004). The Network Dynamics Hypothesis: How Channel Networks Structure Riverine Habitats. *BioScience,* 54**,** 413-427.

BIGELOW, P., BENDA, L. and PEARCE, S. (2016). Delineating incised stream sediment sources within a San Francisco Bay tributary basin. *Earth Surface Dynamics,* 4.

BOARDMAN, J. (2003). Soil erosion and flooding on the eastern South Downs, southern England, 1976–2001. *Transactions of the Institute of British Geographers,* 28**,** 176-196.

BOORMAN, D., HOLLIS, J. M. and LILLY, A. (1995). *Hydrology of soil types: a hydrologically-based classification of the soils of United Kingdom*, Institute of Hydrology.

BORCARD, D., GILLET, F. and LEGENDRE, P. (2011). *Numerical Ecology with R,* New York, NY, UNITED STATES, Springer.

BORJA, Á., GALPARSORO, I., SOLAUN, O., MUXIKA, I., TELLO, E. M., URIARTE, A. and VALENCIA, V. (2006). The European Water Framework Directive and the DPSIR, a methodological approach to assess the risk of failing to achieve good ecological status. *Estuarine, Coastal and Shelf Science,* 66**,** 84-96.

BOWES, M. J., ARMSTRONG, L. K., HARMAN, S. A., WICKHAM, H. D., NICHOLLS, D. J. E., SCARLETT, P. M., ROBERTS, C., JARVIE, H. P., OLD, G. H., GOZZARD, E., BACHILLER-JARENO, N. and READ, D. S. (2018). Weekly water quality monitoring data for the River Thames (UK) and its major tributaries (2009–2013): the Thames Initiative research platform. *Earth Syst. Sci. Data,* 10**,** 1637-1653.

BRACK, W., DULIO, V., ÅGERSTRAND, M., ALLAN, I., ALTENBURGER, R., BRINKMANN, M., BUNKE, D., BURGESS, R. M., COUSINS, I., ESCHER, B. I., HERNÁNDEZ, F. J., HEWITT, L. M., HILSCHEROVÁ, K., HOLLENDER, J., HOLLERT, H., KASE, R., KLAUER, B., LINDIM, C., HERRÁEZ, D. L., MIÈGE, C., MUNTHE, J., O'TOOLE, S., POSTHUMA, L., RÜDEL, H., SCHÄFER, R. B., SENGL, M., SMEDES, F., VAN DE MEENT, D., VAN DEN BRINK, P. J., VAN GILS, J., VAN WEZEL, A. P., VETHAAK, A. D., VERMEIRSSEN, E., VON DER OHE, P. C. and VRANA, B. (2017). Towards the review of the European Union Water Framework Directive: Recommendations for more efficient assessment and management of chemical contamination in European surface water resources. *Science of The Total Environment,* 576**,** 720-737.

BRICE, J. C. (1964). *Channel patterns and terraces of the Loup Rivers in Nebraska*, US Government Printing Office.

BRIERLEY, G. J. and FRYIRS, K. (2000). River styles, a geomorphic approach to catchment characterization: Implications for river rehabilitation in Bega catchment, New South Wales, Australia. *Environmental Management,* 25**,** 661-679.

BRIERLEY, G. J. and FRYIRS, K. A. (2013). *Geomorphology and river management: applications of the river styles framework*, John Wiley and Sons.

BRIERLEY, G. J. (1996). Channel morphology and element assemblages: a constructivist approach to facies modelling. *Advances in fluvial dynamics and stratigraphy***,** 263-298.

BROOKES, A. (1994). River channel change. *The rivers handbook: hydrological and ecological principles,* 2**,** 55-75.

BROOKS, A. P. and BRIERLEY, G. J. (1997). Geomorphic responses of lower Bega River to catchment disturbance, 1851–1926. *Geomorphology,* 18**,** 291-304.

BROOKS, S. J., FITCH, B., DAVY-BOWKER, J. and CODESAL, S. A. (2019). Anglers' Riverfly Monitoring Initiative (ARMI): A UK-wide citizen science project for water quality assessment. *Freshwater Science,* 38**,** 270-280.

BROWN, A. G., LESPEZ, L., SEAR, D. A., MACAIRE, J.-J., HOUBEN, P., KLIMEK, K., BRAZIER, R. E., VAN OOST, K. and PEARS, B. (2018). Natural vs anthropogenic streams in Europe: history, ecology and implications for restoration, river-rewilding and riverine ecosystem services. *Earth-science reviews,* 180**,** 185-205.

BROWN, A. G., TOOTH, S., BULLARD, J. E., THOMAS, D. S., CHIVERRELL, R. C., PLATER, A. J., MURTON, J., THORNDYCRAFT, V. R., TAROLLI, P. and ROSE, J. (2017). The geomorphology of the Anthropocene: emergence, status and implications. *Earth Surface Processes and Landforms,* 42**,** 71-90.

BROWN, A. G., TOOTH, S., CHIVERRELL, R. C., ROSE, J., THOMAS, D. S., WAINWRIGHT, J., BULLARD, J. E., THORNDYCRAFT, V. R., AALTO, R. and DOWNS, P. (2013). The Anthropocene: is there a geomorphological case? *Earth Surface Processes and Landforms,* 38**,** 431-434.

BUFFINGTON, J. and MONTGOMERY, D. (2013). Geomorphic classification of rivers. *In: Shroder, J.; Wohl, E., ed. Treatise on Geomorphology; Fluvial Geomorphology, Vol. 9. San Diego, CA: Academic Press. p. 730-767.***,** 730-767.

BURT, T., HOWDEN, N., WORRALL, F. and WHELAN, M. (2010). Long-term monitoring of river water nitrate: how much data do we need? *Journal of Environmental Monitoring,* 12**,** 71-79.

CALDWELL, P., SUN, G., MCNULTY, S., COHEN, E. and MYERS, J. M. (2012). Impacts of impervious cover, water withdrawals, and climate change on river flows in the conterminous US. *Hydrol. Earth Syst. Sci. 16: 2839–2857,* 16**,** 2839-2857.

CALLISTO, M., MORENO, P. and BARBOSA, F. A. R. (2001). Habitat diversity and benthic functional trophic groups at Serra do Cipó, Southeast Brazil. *Revista Brasileira de Biologia,* 61**,** 259-266.

CARVALHO, L., CORTES, R. and BORDALO, A. A. (2011). Evaluation of the ecological status of an impaired watershed by using a multi-index approach. *Environmental Monitoring and Assessment,* 174**,** 493-508.

CASTRO, J. M. and THORNE, C. R. (2019). The stream evolution triangle: Integrating geology, hydrology, and biology. *River Research and Applications,* 35**,** 315-326.

CEH. (2019). Flood estimation handbook [Online]. Available at: https://www.ceh.ac.uk/services/flood-estimation-handbook. (Accessed 3rd September 2019).

CHAMPKIN, J., COPP, G., SAYER, C., CLILVERD, H., GEORGE, L., VILIZZI, L., GODARD, M., CLARKE, J. and WALKER, A. (2018). Responses of fishes and lampreys to the re-creation of meanders in a small English chalk stream. *River research and applications,* 34**,** 34-43.

CHARLTON, R. (2007). *Fundamentals of fluvial geomorphology*, Routledge.

CHAZDON, R. L., PERES, C. A., DENT, D., SHEIL, D., LUGO, A. E., LAMB, D., STORK, N. E. and MILLER, S. E. (2009). The Potential for Species Conservation in Tropical Secondary Forests. *Conservation Biology,* 23**,** 1406-1417.

CIVAN, A., WORRALL, F., JARVIE, H. P., HOWDEN, N. J. K. and BURT, T. P. (2018). Forty-year trends in the flux and concentration of phosphorus in British rivers. *Journal of Hydrology,* 558**,** 314-327.

CLUER, B. and THORNE, C. (2014). A stream evolution model integrating habitat and ecosystem benefits. *River Research and Applications,* 30**,** 135-154.

COCHERO, J., CORTELEZZI, A., TARDA, A. S. and GÓMEZ, N. (2016). An index to evaluate the fluvial habitat degradation in lowland urban streams. *Ecological indicators,* 71**,** 134-144.

COFFIN, A. W. (2007). From roadkill to road ecology: a review of the ecological effects of roads. *Journal of transport Geography,* 15**,** 396-406.

COUPER, P. R. and MADDOCK, I. P. (2001). Subaerial river bank erosion processes and their interaction with other bank erosion mechanisms on the River Arrow, Warwickshire, UK. *Earth Surface Processes and Landforms: The Journal of the British Geomorphological Research Group,* 26**,** 631-646.

CRUTZEN, P. J. (2006). The “anthropocene”. *Earth system science in the anthropocene.* Springer.

DAPPORTO, S., RINALDI, M., CASAGLI, N. and VANNOCCI, P. (2003). Mechanisms of riverbank failure along the Arno River, Central Italy. *Earth Surface Processes and Landforms: The Journal of the British Geomorphological Research Group,* 28**,** 1303-1323.

DAVENPORT, A. J., GURNELL, A. M. and ARMITAGE, P. D. (2004). Habitat survey and classification of urban rivers. *River Research and Applications,* 20**,** 687-704.

DEVIA, G. K., GANASRI, B. and DWARAKISH, G. (2015). A review on hydrological models. *Aquatic Procedia,* 4**,** 1001-1007.

DODDS, W. K. and SMITH, V. H. (2016). Nitrogen, phosphorus, and eutrophication in streams. *Inland Waters,* 6**,** 155-164.

DOICHINOVA, V., ZHIYANSKI, M. and HURSTHOUSE, A. (2006). Impact of urbanisation on soil characteristics. *Environmental Chemistry Letters,* 3**,** 160-163.

DOWNS, P. and GREGORY, K. (2014). *River channel management: towards sustainable catchment hydrosystems*, Routledge.

DOWNS, P. W., DUSTERHOFF, S. R. and SEARS, W. A. (2013). Reach-scale channel sensitivity to multiple human activities and natural events: Lower Santa Clara River, California, USA. *Geomorphology,* 189**,** 121-134.

DOXIADIS, C. A. (1968). Ekistics; an introduction to the science of human settlements.

DUFOUR, S. and PIÉGAY, H. (2009). From the myth of a lost paradise to targeted river restoration: forget natural references and focus on human benefits. *River research and applications,* 25**,** 568-581.

DUGDALE, S. J., BERGERON, N. E. and ST-HILAIRE, A. (2013). Temporal variability of thermal refuges and water temperature patterns in an Atlantic salmon river. *Remote Sensing of Environment,* 136**,** 358-373.

ECUWW. (2019). Urban Wastewater Treatment Directive treatment map [Online]. Available at: https://uwwtd.eu/United-Kingdom/uwwtps/treatment. (Accessed 3rd September 2019).

EEA. (2019). Urban wastewater treatment map [Online]. Available at: https://www.eea.europa.eu/themes/water/european-waters/water-use-and-environmental-pressures/uwwtd/interactive-maps/urban-waste-water-treatment-maps. (Accessed 3rd September 2019).

ELLIS, E. C. (2011). Anthropogenic transformation of the terrestrial biosphere. *Philosophical Transactions of the Royal Society A: Mathematical, Physical and Engineering Sciences,* 369**,** 1010-1035.

ELLIS, E. C., KLEIN GOLDEWIJK, K., SIEBERT, S., LIGHTMAN, D. and RAMANKUTTY, N. (2010). Anthropogenic transformation of the biomes, 1700 to 2000. *Global ecology and biogeography,* 19**,** 589-606.

Environment Agency. (2019). Water Quality archive documentation [Online]. Available at: https://environment.data.gov.uk/water-quality/view/doc/reference. (Accessed 3rd September 2019).

EVERALL, N. C., JOHNSON, M. F., WOOD, P., FARMER, A., WILBY, R. L. and MEASHAM, N. (2017). Comparability of macroinvertebrate biomonitoring indices of river health derived from semi-quantitative and quantitative methodologies. *Ecological Indicators,* 78**,** 437-448.

EVERARD, M. and POWELL, A. (2002). Rivers as living systems. *Aquatic Conservation: Marine and Freshwater Ecosystems,* 12**,** 329-337.

EXTENCE, A. C., P. CHADD, R., ENGLAND, J., J. DUNBAR, M., J. WOOD, P. and D. TAYLOR, E. (2013). The assessment of fine sediment accumulation in rivers using macro-invertebrate community response. *River Research and Applications,* 29**,** 17-55.

EXTENCE, C., BALBI, D. and CHADD, R. (1999). River flow indexing using British benthic macroinvertebrates: a framework for setting hydroecological objectives. *Regulated Rivers: Research and Management: An International Journal Devoted to River Research and Management,* 15**,** 545-574.

FALAGÁN, C., SMAIL, I., GRAIL, B. M. and JOHNSON, D. B. (2016). Microbial communities in passive remediation systems at three abandoned coal mine sites in the United Kingdom. Proceedings of the IMWA 2016 Symposium: Leipzig, Germany.

FENN, M. (2018). River Ecology and Urban Pollution. *Urban Pollution: Science and Management***,** 173-186.

FERNÁNDEZ, D., BARQUÍN, J. and RAVEN, P. J. (2011). A review of river habitat characterisation methods: Indices vs.characterisation protocols. *Limnetica,* 30**,** 217-234.

FLORSHEIM, J. L., MOUNT, J. F. and CHIN, A. (2008). Bank erosion as a desirable attribute of rivers. *BioScience,* 58**,** 519-529.

FOX, P. J., NAURA, M. and SCARLETT, P. (1998). An account of the derivation and testing of a standard field method, River Habitat Survey. *Aquatic Conservation: Marine and Freshwater Ecosystems,* 8**,** 455-475.

FRECHETTE, D. M., DUGDALE, S. J., DODSON, J. J. and BERGERON, N. E. (2018). Understanding summertime thermal refuge use by adult Atlantic salmon using remote sensing, river temperature monitoring, and acoustic telemetry. *Canadian Journal of Fisheries and Aquatic Sciences,* 75**,** 1999-2010.

FRISSELL, C. A., LISS, W. J., WARREN, C. E. and HURLEY, M. D. (1986). A hierarchical framework for stream habitat classification: viewing streams in a watershed context. *Environmental management,* 10**,** 199-214.

GEORGE, S. D., BALDIGO, B. P., LAWRENCE, G. B. and FULLER, R. L. (2018). Effects of watershed and in-stream liming on macroinvertebrate communities in acidified tributaries to an Adirondack lake. *Ecological indicators,* 85**,** 1058-1067.

GODWIN, K., HAFNER, S. and BUFF, M. (2003). Long-term trends in sodium and chloride in the Mohawk River, New York: the effect of fifty years of road-salt application. *Environmental pollution,* 124**,** 273-281.

GOUDIE, A. S., VILES, H.A. (2016). *Geomorphology in the anthropocene,* Oxford, Oxford University Press.

GRACIA, L., BACON, J., CARLOS, C. (2018). Modelling the fate and transport of faecal bacteria from sewage overflows: The Dart Estuary case study. *25th TELEMAC-MASCARET User Conference 2018* Norwich: HENRY.

GREENWOOD, P., BAUMANN, P., PULLEY, S. and KUHN, N. J. (2018). The invasive alien plant, Impatiens glandulifera (Himalayan Balsam), and increased soil erosion: causation or association? Case studies from a river system in Switzerland and the UK. *Journal of soils and sediments,* 18**,** 3463-3477.

GRIFFITH, D. A. and AMRHEIN, C. (1997). *Multivariate statistical analysis for geographers*.

GUARESCHI, S. and WOOD, P. J. (2019). Taxonomic changes and non-native species: An overview of constraints and new challenges for macroinvertebrate-based indices calculation in river ecosystems. *Science of the Total Environment,* 660**,** 40-46.

GUARESCHI, S., LAINI, A. and SÁNCHEZ-MONTOYA, M. M. (2017). How do low-abundance taxa affect river biomonitoring? Exploring the response of different macroinvertebrate-based indices. *Journal of Limnology*.

HADDADCHI, A., BOOKER, D. J. and MEASURES, R. J. (2018). Predicting river bed substrate cover proportions across New Zealand. *Catena,* 163**,** 130-146.

HAMMER, T. R. (1972). Stream channel enlargement due to urbanization. *Water Resources Research,* 8**,** 1530-1540.

HERING, D., BORJA, A., CARSTENSEN, J., CARVALHO, L., ELLIOTT, M., FELD, C. K., HEISKANEN, A.-S., JOHNSON, R. K., MOE, J. and PONT, D. (2010). The European Water Framework Directive at the age of 10: a critical review of the achievements with recommendations for the future. *Science of the total Environment,* 408**,** 4007-4019.

HEY, R. D. and THORNE, C. R. (1983). Accuracy of surface samples from gravel bed material. *Journal of Hydraulic Engineering,* 109**,** 842-851.

HEY, R. D. (1979). Dynamic process-response model of river channel development. *Earth Surface Processes,* 4**,** 59-72.

HICKIN, E. J. (1984). Vegetation and river channel dynamics. *Canadian Geographer/Le Géographe canadien,* 28**,** 111-126.

HOBBS, R. J., ARICO, S., ARONSON, J., BARON, J. S., BRIDGEWATER, P., CRAMER, V. A., EPSTEIN, P. R., EWEL, J. J., KLINK, C. A. and LUGO, A. E. (2006). Novel ecosystems: theoretical and management aspects of the new ecological world order. *Global ecology and biogeography,* 15**,** 1-7.

HOBBS, R. J., HIGGS, E. and HARRIS, J. A. (2009). Novel ecosystems: implications for conservation and restoration. *Trends in Ecology and Evolution,* 24**,** 599-605.

HOBBS, R. J., HIGGS, E., HALL, C. M., BRIDGEWATER, P., CHAPIN III, F. S., ELLIS, E. C., EWEL, J. J., HALLETT, L. M., HARRIS, J. and HULVEY, K. B. (2014). Managing the whole landscape: historical, hybrid, and novel ecosystems. *Frontiers in Ecology and the Environment,* 12**,** 557-564.

HOLMES, N., BOON, P. and ROWELL, T. (1998). A revised classification system for British rivers based on their aquatic plant communities. *Aquatic Conservation: marine and freshwater ecosystems,* 8**,** 555-578.

HORNE, A., KAUR, S., SZEMIS, J., COSTA, A., WEBB, J. A., NATHAN, R., STEWARDSON, M., LOWE, L. and BOLAND, N. (2017). Using optimization to develop a "designer" environmental flow regime. *Environmental modelling and software,* 88**,** 188-199.

HOWDEN, N. J. K., BURT, T. P., WORRALL, F., MATHIAS, S. A. and WHELAN, M. J. (2013). Farming for Water Quality: Balancing Food Security and Nitrate Pollution in UK River Basins. *Annals of the Association of American Geographers,* 103**,** 397-407.

JARVIE, H. P., SMITH, D. R., NORTON, L. R., EDWARDS, F. K., BOWES, M. J., KING, S. M., SCARLETT, P., DAVIES, S., DILS, R. M. and BACHILLER-JARENO, N. (2018). Phosphorus and nitrogen limitation and impairment of headwater streams relative to rivers in Great Britain: A national perspective on eutrophication. *Science of The Total Environment,* 621**,** 849-862.

JELIC, A., GROS, M., GINEBREDA, A., CESPEDES-SÁNCHEZ, R., VENTURA, F., PETROVIC, M. and BARCELO, D. (2011). Occurrence, partition and removal of pharmaceuticals in sewage water and sludge during wastewater treatment. *Water Research,* 45**,** 1165-1176.

JOHNSON, M. F. and WILBY, R. L. (2015). Seeing the landscape for the trees: Metrics to guide riparian shade management in river catchments. *Water Resources Research,* 51**,** 3754-3769.

JOHNSON, M. F., RICE, S. P. and REID, I. (2011). Increase in coarse sediment transport associated with disturbance of gravel river beds by signal crayfish (Pacifastacus leniusculus). *Earth Surface Processes and Landforms,* 36**,** 1680-1692.

JOHNSON, M. F., THORNE, C. R., CASTRO, J. M., KONDOLF, G. M., MAZZACANO, C. S., ROOD, S. B. and WESTBROOK, C. (2019). Biomic river restoration: A new focus for river management. *River Research and Applications*.

JOHNSON, M. L., PASTERNACK, G., FLORSHEIM, J., WERNER, I., SMITH, T. B., BOWEN, L., TURNER, M., VIERS, J., STEINMETZ, J. and CONSTANTINE, J. (2002). North coast river loading study road crossing on small streams volume i. status of salmonids in the watershed. California Department of the Environment: CTSW-RT-02-040. Sacramento, CA, Caltrans,.

JONES, J., MURPHY, J., COLLINS, A., SEAR, D., NADEN, P. and ARMITAGE, P. (2012). The impact of fine sediment on macro-invertebrates. *River Research and Applications,* 28**,** 1055-1071.

JONES, M. E. (2000). Road upgrade, road mortality and remedial measures: impacts on a population of eastern quolls and Tasmanian devils. *Wildlife research,* 27**,** 289-296.

JUNK, W. J., BAYLEY, P. B. and SPARKS, R. E. (1989). The flood pulse concept in river-floodplain systems. *Canadian special publication of fisheries and aquatic sciences,* 106**,** 110-127.

KALLIS, G. and BUTLER, D. (2001). The EU water framework directive: measures and implications. *Water Policy,* 3**,** 125-142.

KANNEL, P. R., LEE, S., LEE, Y.-S., KANEL, S. R. and KHAN, S. P. (2007). Application of water quality indices and dissolved oxygen as indicators for river water classification and urban impact assessment. *Environmental monitoring and assessment,* 132**,** 93-110.

KNIGHTON, D. (2014). *Fluvial forms and processes: a new perspective*, Routledge.

KONDOLF, G. M. (1997). APPLICATION OF THE PEBBLE COUNT NOTES ON PURPOSE, METHOD, AND VARIANTS1. *JAWRA Journal of the American Water Resources Association,* 33**,** 79-87.

KONDOLF, G. M., SMELTZER, M. W. and RAILSBACK, S. F. (2001). Design and performance of a channel reconstruction project in a coastal California gravel-bed stream. *Environmental Management,* 28**,** 761-776.

KRAJENBRINK, H. J., ACREMAN, M., DUNBAR, M. J., HANNAH, D. M., LAIZÉ, C. L. R. and WOOD, P. J. (2019). Macroinvertebrate community responses to river impoundment at multiple spatial scales. *Science of The Total Environment,* 650**,** 2648-2656.

LACEY, R. J., NEARY, V. S., LIAO, J. C., ENDERS, E. C. and TRITICO, H. M. (2012). The IPOS framework: linking fish swimming performance in altered flows from laboratory experiments to rivers. *River Research and Applications,* 28**,** 429-443.

LAINI, A., BOLPAGNI, R., CANCELLARIO, T., GUARESCHI, S., RACCHETTI, E. and VIAROLI, P. (2018). Testing the response of macroinvertebrate communities and biomonitoring indices under multiple stressors in a lowland regulated river. *Ecological Indicators,* 90**,** 47-53.

LAIOLO, P., GABELLANI, S., CAMPO, L., SILVESTRO, F., DELOGU, F., RUDARI, R., PULVIRENTI, L., BONI, G., FASCETTI, F. and PIERDICCA, N. (2016). Impact of different satellite soil moisture products on the predictions of a continuous distributed hydrological model. *International journal of applied earth observation and geoinformation,* 48**,** 131-145.

LANE, E. W. (1955). Importance of fluvial morphology in hydraulic engineering. *Proceedings of the American Society of Civil Engineers; v. 81, paper no. 745*.

LEIVESTAD, H. and MUNIZ, I. (1976). Fish kill at low pH in a Norwegian river. *Nature,* 259**,** 391.

LEOPOLD, L. B. and MADDOCK, T. (1953). *The hydraulic geometry of stream channels and some physiographic implications*, US Government Printing Office.

LEOPOLD, L. B. and WOLMAN, M. G. (1957). *River channel patterns: braided, meandering, and straight*, US Government Printing Office.

LEPOM, P., BROWN, B., HANKE, G., LOOS, R., QUEVAUVILLER, P. and WOLLGAST, J. (2009). Needs for reliable analytical methods for monitoring chemical pollutants in surface water under the European Water Framework Directive. *Journal of Chromatography A,* 1216**,** 302-315.

MACKLIN, M., LEWIN, J. and JONES, A. F. (2014). Anthropogenic alluvium: an evidence-based meta-analysis for the UK Holocene. *Anthropocene,* 6**,** 26-38.

MADDOCK, I. (1999). The importance of physical habitat assessment for evaluating river health. *Freshwater biology,* 41**,** 373-391.

MAINSTONE, C. P. and PARR, W. (2002). Phosphorus in rivers — ecology and management. *Science of The Total Environment,* 282-283**,** 25-47.

MALAJ, E., PETER, C., GROTE, M., KÜHNE, R., MONDY, C. P., USSEGLIO-POLATERA, P., BRACK, W. and SCHÄFER, R. B. (2014). Organic chemicals jeopardize the health of freshwater ecosystems on the continental scale. *Proceedings of the National Academy of Sciences,* 111**,** 9549-9554.

MALLIN, M. A., JOHNSON, V. L., ENSIGN, S. H. and MACPHERSON, T. A. (2006). Factors contributing to hypoxia in rivers, lakes, and streams. *Limnology and Oceanography,* 51**,** 690-701.

MARTIN, L. J., QUINN, J. E., ELLIS, E. C., SHAW, M. R., DORNING, M. A., HALLETT, L. M., HELLER, N. E., HOBBS, R. J., KRAFT, C. E. and LAW, E. (2014). Conservation opportunities across the world's anthromes. *Diversity and Distributions,* 20**,** 745-755.

MCGINNITY, P., MILLS, P., ROCHE, W. AND MULLER, M. (2002). WATER FRAMEWORK DIRECTIVE –A Desk Study to Determine a Methodology for the Monitoring of the ‘Morphological Condition’ of Irish Rivers. *Environmental Protection Agency***,** 26.

MILLER, A. M. and GOLLADAY, S. W. (1996). Effects of spates and drying on macroinvertebrate assemblages of an intermittent and a perennial prairie stream. *Journal of the North American Benthological Society,* 15**,** 670-689.

MONTGOMERY, D. R. and BUFfiNGTON, J. M. (1998). Channel processes, classification, and response. *River Ecology and Management: Lessons from the Pacific Coastal Ecoregion, RJ Naiman and RE Bilby (Editors). Springer-Verlag, New York, New York***,** 13-42.

MONTGOMERY, D. R. (2001). Geomorphology, river ecology, and ecosystem management. *Geomorphic processes and riverine habitat,* 4**,** 247-253.

MONTGOMERY, D. R. (2008). Dreams of natural streams. *Science,* 319**,** 291-292.

MOORE, J. W. (2006). Animal ecosystem engineers in streams. *BioScience,* 56, 237-246.

MUSGRAVE, G. (1935). 1801401. The infiltration capacity of soils in relation to the control of surface runoff and erosion. *Agronomy journal,* 27, 336-345.

NADEN, P., BELL, V., CARNELL, E., TOMLINSON, S., DRAGOSITS, U., CHAPLOW, J., MAY, L. and TIPPING, E. (2016). Nutrient fluxes from domestic wastewater: a national-scale historical perspective for the UK 1800–2010. *Science of the Total Environment,* 572, 1471-1484.

NAIMAN, R. J., DECAMPS, H. and MCCLAIN, M. E. (2010). *Riparia: ecology, conservation, and management of streamside communities*, Elsevier.

NAVRATIL, O., BREIL, P., SCHMITT, L., GROSPRÊTRE, L. and ALBERT, M. B. (2013). Hydrogeomorphic adjustments of stream channels disturbed by urban runoff (Yzeron River basin, France). *Journal of Hydrology,* 485, 24-36.

NEGISHI, J., INOUE, M. and NUNOKAWA, M. (2002). Effects of channelisation on stream habitat in relation to a spate and flow refugia for macroinvertebrates in northern Japan. *Freshwater Biology,* 47, 1515-1529.

NEWSON, M. D. and LARGE, A. R. (2006). 'Natural'rivers,'hydromorphological quality'and river restoration: a challenging new agenda for applied fluvial geomorphology. *Earth Surface Processes and Landforms,* 31, 1606-1624.

NEWSON, M. D. and NEWSON, C. L. (2000). Geomorphology, ecology and river channel habitat: mesoscale approaches to basin-scale challenges. *Progress in Physical Geography: Earth and Environment,* 24, 195-217.

NEWSON, M. D. (2002). Geomorphological concepts and tools for sustainable river ecosystem management. *Aquatic Conservation: Marine and Freshwater Ecosystems,* 12, 365-379.

NIXON, S. (1996). *Development and Testing of General Quality Assessment Schemes*, HM Stationery Office.

NORRIS, R. H. and THOMS, M. C. (1999). What is river health? *Freshwater Biology,* 41, 197-209.

NOTEBAERT, B., BROOTHAERTS, N. and VERSTRAETEN, G. (2018). Evidence of anthropogenic tipping points in fluvial dynamics in Europe. *Global and Planetary Change,* 164, 27-38.

OBERHOLSTER, P. J., BOTHA, A.-M., HILL, L. and STRYDOM, W. F. (2017). River catchment responses to anthropogenic acidification in relationship with sewage effluent: an ecotoxicology screening application. *Chemosphere,* 189, 407-417.

OLDEN, J. D. and POFF, N. L. (2003). Redundancy and the choice of hydrologic indices for characterizing streamflow regimes. *River Research and Applications,* 19, 101-121.

OLSHANSKY, R. B. and KARTEZ, J. D. (1998). Managing land use to build resilience. *Cooperating with nature: Confronting natural hazards with land-use planning for sustainable communities*, 167-202.

PAISLEY, M., TRIGG, D. and WALLEY, W. (2014). Revision of the biological monitoring working party (bmwp) score system: derivation of present-only and abundance-related scores from field data. *River research and applications,* 30, 887-904.

PALMER, M. A., MENNINGER, H. L. and BERNHARDT, E. (2010). River restoration, habitat heterogeneity and biodiversity: a failure of theory or practice? *Freshwater Biology,* 55, 205-222.

PETTS, G. E. (1984). *Impounded rivers: perspectives for ecological management*, Wiley.

PILCHER, M. W., COPP, G. H. and SZOMOLAI, V. (2004). A comparison of adjacent natural and channelised stretches of a lowland river. *BIOLOGIA-BRATISLAVA-,* 59, 669-674.

POEPPL, R. E., KEESSTRA, S. D. and HEIN, T. (2015). The geomorphic legacy of small dams—An Austrian study. *Anthropocene,* 10, 43-55.

POESEN, J. (1984). The influence of slope angle on infiltration rate and Hortonian overland flow. *Zeitschrift für Geomorpholgie, Supplement Band,* 49, 117-131.

POFF, N. L. and WARD, J. (1990). Physical habitat template of lotic systems: recovery in the context of historical pattern of spatiotemporal heterogeneity. *Environmental management,* 14, 629.

POFF, N. L., RICHTER, B. D., ARTHINGTON, A. H., BUNN, S. E., NAIMAN, R. J., KENDY, E., ACREMAN, M., APSE, C., BLEDSOE, B. P. and FREEMAN, M. C. (2010). The ecological limits of hydrologic alteration (ELOHA): a new framework for developing regional environmental flow standards. *Freshwater Biology,* 55, 147-170.

POLLOCK, M. M., BEECHIE, T. J., WHEATON, J. M., JORDAN, C. E., BOUWES, N., WEBER, N. and VOLK, C. (2014). Using beaver dams to restore incised stream ecosystems. *Bioscience,* 64, 279-290.

PRIOR, J. (2016). Urban river design and aesthetics: a river restoration case study from the UK. *Journal of Urban Design,* 21, 512-529.

RAVEN, P., HOLMES, N., DAWSON, F. and EVERARD, M. (1998). Quality assessment using river habitat survey data. *Aquatic conservation: marine and freshwater ecosystems,* 8, 477-499.

REVELLE, W. (1979). Hierarchical Cluster Analysis And The Internal Structure Of Tests. *Multivariate Behavioral Research,* 14, 57-74.

REYJOL, Y., ARGILLIER, C., BONNE, W., BORJA, A., BUIJSE, A. D., CARDOSO, A. C., DAUFRESNE, M., KERNAN, M., FERREIRA, M. T. and POIKANE, S. (2014). Assessing the ecological status in the context of the European Water Framework Directive: where do we go now? *Science of the Total Environment,* 497, 332-344.

ROBERT, A. (2014). *River processes: an introduction to fluvial dynamics*, Routledge.

ROBERTS, E. J. and COOPER, R. J. (2018). Riverbed sediments buffer phosphorus concentrations downstream of sewage treatment works across the River Wensum catchment, UK. *Journal of Soils and Sediments,* 18**,** 2107-2116.

ROSGEN, D. L. (1994). A classification of natural rivers. *Catena,* 22**,** 169-199.

RUDOLF, A., AHUMADA, R. and PE'REZ, C. (2002). Dissolved oxygen content as an index of water quality in San Vicente Bay, Chile (36 degrees, 450S). *Environmental Monitoring and Assessment,* 78.

SÁNCHEZ, E., COLMENAREJO, M. F., VICENTE, J., RUBIO, A., GARCÍA, M. G., TRAVIESO, L. and BORJA, R. (2007). Use of the water quality index and dissolved oxygen deficit as simple indicators of watersheds pollution. *Ecological indicators,* 7**,** 315-328.

SAPKALE, J. B., KADAM, Y. U., JADHAV, I. A. and KAMBLE, S. S. (2016). River in Planform and Variation in Sinuosity Index: A Study of Dhamni River, Kolhapur (Maharashtra), India. *Int. J. Sci. Eng. Res,* 7**,** 863-867.

SCANLON, B. R., HEALY, R. W. and COOK, P. G. (2002). Choosing appropriate techniques for quantifying groundwater recharge. *Hydrogeology journal,* 10**,** 18-39.

SCHUMM, S. A. 1977. The fluvial system.

SCHUMM, S. A., HARVEY, M. D. and WATSON, C. C. (1984). *Incised channels: morphology, dynamics, and control*, Water Resources Publications.

SERLET, A. J., GURNELL, A. M., ZOLEZZI, G., WHARTON, G., BELLEUDY, P. and JOURDAIN, C. (2018). Biomorphodynamics of alternate bars in a channelized, regulated river: An integrated historical and modelling analysis. *Earth Surface Processes and Landforms,* 43**,** 1739-1756.

SHAFER, L. L., JAMES, J. W., RATH, R. D. and EUBANK, J. (2009). Waste water treatment method. Google Patents.

SHREVE, R. L. (1966). Statistical law of stream numbers. *The Journal of Geology,* 74**,** 17-37.

SIMON, A. and COLLISON, A. J. C. (2002). Quantifying the mechanical and hydrologic effects of riparian vegetation on streambank stability. *Earth Surface Processes and Landforms,* 27**,** 527-546.

SIMON, A., CASTRO, J. and RINALDI, M. (2016). Channel form and adjustment: characterization, measurement, interpretation and analysis. *Tools in Fluvial Geomorphology***,** 233-259.

SIMON, A., DOYLE, M., KONDOLF, M., SHIELDS JR, F., RHOADS, B. and MCPHILLIPS, M. (2007). Critical Evaluation of How the Rosgen Classification and Associated "Natural Channel Design" Methods Fail to Integrate and Quantify Fluvial Processes and

Channel Response 1. *JAWRA Journal of the American Water Resources Association,* 43**,** 1117-1131.

SMITH, V. H. (2003). Eutrophication of freshwater and coastal marine ecosystems a global problem. *Environmental Science and Pollution Research,* 10**,** 126-139.

STEFFEN, W., RICHARDSON, K., ROCKSTRÖM, J., CORNELL, S. E., FETZER, I., BENNETT, E. M., BIGGS, R., CARPENTER, S. R., DE VRIES, W. and DE WIT, C. A. (2015). Planetary boundaries: Guiding human development on a changing planet. *Science,* 347**,** 1259855.

STERLING, E. J., BURKS, R. L., LINDER, J., LANGEN, T., FERNANDEZ, D. S., RUBY, D. and BYNUM, N. (2018). Why is Biodiversity Important? An Oral Communication Exercise. *LESSONS IN CONSERVATION***,** 11.

STOCK, J. D., MONTGOMERY, D. R., COLLINS, B. D., DIETRICH, W. E. and SKLAR, L. (2005). Field measurements of incision rates following bedrock exposure: Implications for process controls on the long profiles of valleys cut by rivers and debris flows. *Geological Society of America Bulletin,* 117**,** 174-194.

STOTT, P. (2016). How climate change affects extreme weather events. *Science,* 352**,** 1517-1518.

STRAHLER, A. N. (1957). Quantitative analysis of watershed geomorphology. *Eos, Transactions American Geophysical Union,* 38**,** 913-920.

TAUBE, N., RYAN, M. C., HE, J. and VALEO, C. (2019). Phosphorus and nitrogen storage, partitioning, and export in a large gravel bed river. *Science of The Total Environment,* 657**,** 717-730.

TAYLOR, M. P., MACKLIN, M. G. and HUDSON-EDWARDS, K. (2000). River sedimentation and fluvial response to Holocene environmental change in the Yorkshire Ouse Basin, northern England. *The Holocene,* 10**,** 201-212.

THARME, R. E. (2003). A global perspective on environmental flow assessment: emerging trends in the development and application of environmental flow methodologies for rivers. *River research and applications,* 19**,** 397-441.

THOMSON, J. R., TAYLOR, M. P. and BRIERLEY, G. J. (2004). Are River Styles ecologically meaningful? A test of the ecological significance of a geomorphic river characterization scheme. *Aquatic Conservation: Marine and Freshwater Ecosystems,* 14**,** 25-48.

THORNE, C. R. (1990). *Effects of vegetation on riverbank erosion and stability*.

THORNE, C. R. (1997). Channel types and morphological classification. *Applied fluvial geomorphology for river engineering and management***,** 175-222.

THORNE, C. R. (1998). *Stream Reconnaissance Handbook: Geomorphological investigation and analysis of river channels,* Chichester, John Wiley and Sons

TOCKNER, K., MALARD, F. and WARD, J. (2000). An extension of the flood pulse concept. *Hydrological processes,* 14**,** 2861-2883.

TRIMBLE, S. W. (2009). Fluvial processes, morphology and sediment budgets in the Coon Creek Basin, WI, USA, 1975–1993. *Geomorphology,* 108**,** 8-23.

TURLEY, M. D., BILOTTA, G. S., EXTENCE, C. A. and BRAZIER, R. E. (2014). Evaluation of a fine sediment biomonitoring tool across a wide range of temperate rivers and streams. *Freshwater biology,* 59**,** 2268-2277.

UK Technical advisory group. (2014). Updated recommendations on environmental standards: River basin management [online]. Available at: http://www.wfduk.org/sites/default/files/Media/Environmental%20standards/UKTAG%20Environmental%20Standards%20Phase%203%20Final%20Report%2004112013.pdf. (Accessed on: 10th July 2019).

VANNOTE, R. L., MINSHALL, G. W., CUMMINS, K. W., SEDELL, J. R. and CUSHING, C. E. (1980). The river continuum concept. *Canadian journal of fisheries and aquatic sciences,* 37**,** 130-137.

VAUGHAN, I. P. and ORMEROD, S. J. (2005). METHODOLOGICAL INSIGHTS: Increasing the value of principal components analysis for simplifying ecological data: a case study with rivers and river birds. *Journal of Applied Ecology,* 42**,** 487-497.

WALLING, D. (2005). Tracing suspended sediment sources in catchments and river systems. *Science of the total environment,* 344**,** 159-184.

WALTER, R. C. and MERRITTS, D. J. (2008). Natural streams and the legacy of water-powered mills. *Science,* 319**,** 299-304.

WANG, J., LIU, Q., HOU, Y., QIN, W., LESSCHEN, J. P., ZHANG, F. and OENEMA, O. (2018). International trade of animal feed: its relationships with livestock density and N and P balances at country level. *Nutrient Cycling in Agroecosystems,* 110**,** 197-211.

WARD. and TOCKNER. (2001). Biodiversity: towards a unifying theme for river ecology. *Freshwater Biology,* 46**,** 807-819.

WARD, J. V., TOCKNER, K., UEHLINGER, U. and MALARD, F. (2001). Understanding natural patterns and processes in river corridors as the basis for effective river restoration. *Regulated Rivers: Research and Management: An International Journal Devoted to River Research and Management,* 17**,** 311-323.

WESTWOOD, C. G., ENGLAND, J., DUNBAR, M. J., HOLMES, N. T., LEEMING, D. J. and HAMMOND, D. (2017). An approach to setting ecological flow thresholds for southern English chalk streams. *Water and environment journal,* 31**,** 528-536.

WILBY, R. L., JOHNSON, M. F. and TOONE, J. A. (2014). Nocturnal river water temperatures: Spatial and temporal variations. *Science of The Total Environment,* 482-483**,** 157-173.

WILLIAMS, G. P. and WOLMAN, M. G. (1984). Downstream effects of dams on alluvial rivers.

WILLIAMS, G. P. (1978). Bank-full discharge of rivers. *Water resources research,* 14**,** 1141-1154.

WILLIAMS, R. J., WHITE, C., HARROW, M. L. and NEAL, C. (2000). Temporal and small-scale spatial variations of dissolved oxygen in the Rivers Thames, Pang and Kennet, UK. *Science of the total environment,* 251**,** 497-510.

WINTERBOURN, M., MCDIFFETT, W. and EPPLEY, S. (2000). Aluminium and iron burdens of aquatic biota in New Zealand streams contaminated by acid mine drainage: effects of trophic level. *Science of the total Environment,* 254**,** 45-54.

WOHL, E. (2017). The significance of small streams. *Frontiers of Earth Science,* 11**,** 447-456.

WOHL, E., LANE, S. N. and WILCOX, A. C. (2015). The science and practice of river restoration. *Water Resources Research,* 51**,** 5974-5997.

WOLMAN, M. G. and MILLER, J. P. (1960). Magnitude and frequency of forces in geomorphic processes. *The Journal of Geology,* 68**,** 54-74.

WOLMAN, M. G. (1954). A method of sampling coarse river-bed material. *EOS, Transactions American Geophysical Union,* 35**,** 951-956.

YOUNGER, P. and ADAMS, R. (1999). Predicting mine water rebound.

ZHOU, W. S., HAO, K. R., JIANG, C. L., CHEN, L., TANG, X. S. and CAI, X. (2019). A New Cross Clustering Algorithm for Improving Performance of Supervised Learning. *Ieee Access,* 7**,** 56713-56723.

Trimble navigation. (2012). Handheld Platform for MobileMapper 120, ProMark 120 and ProMark 220 [Online]. Available at: https://surveyinstrumentsales.com/PDFs/Manuals/Spectra-Precision-ProMark-120-Manual.pdf. (Accessed on 30th August 2019).

Scottish Environmental Protection Agency. (2015). Natural Flood Management Handbook. Scottish Environmental Protection Agency: Stirling.

Scottish Environmental Protection Agency. (2013). Fluvial audit method statement [Online]. Available at: https://www.sepa.org.uk/media/103075/appendix-a-fluvial-audit-method.pdf. (Accessed on 3rd September 2019).

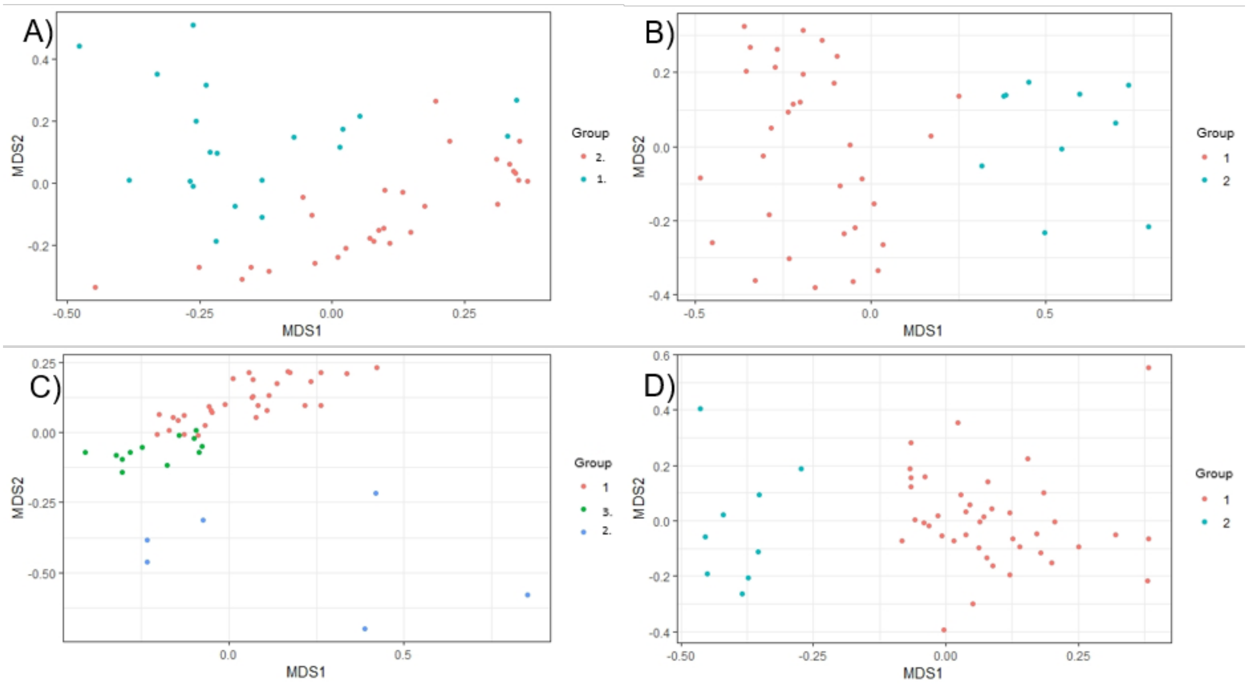
9.0. Appendix

The HMS score for a site is the total of all the component scores in the categories listed below.

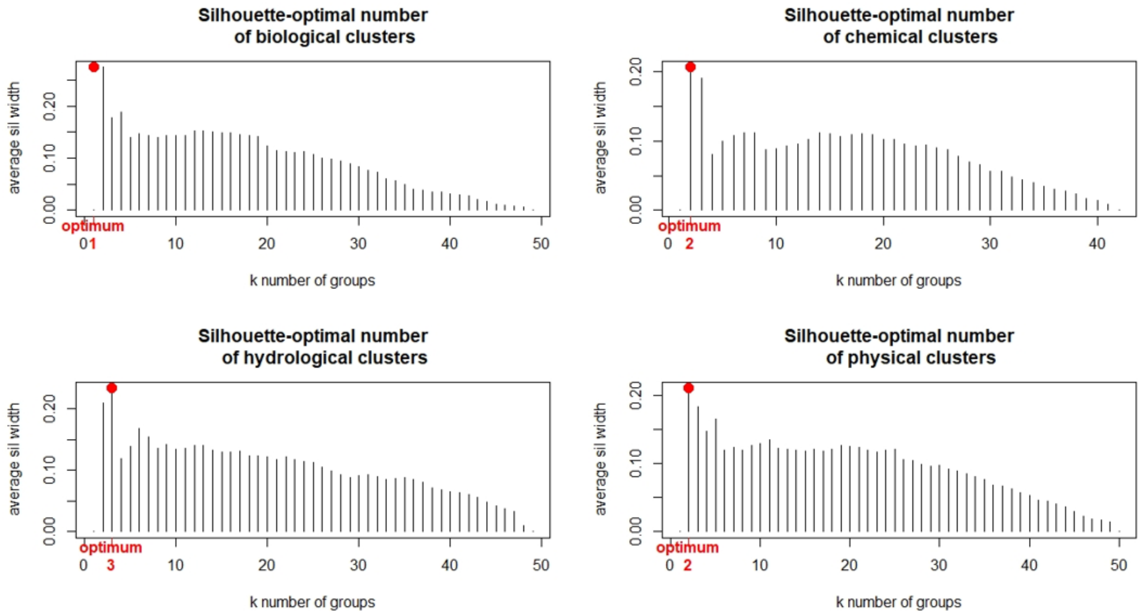
A. Modifications at spot-checks	**Score per spot-check**	
Reinforcement to banks	2	
Reinforcement to bed	2	
Resectioned bank or bed	1	
Two-stage bank modification	1	
Embankment	1	
Culvert	8	
Dam, weir, ford	2	
Bank poached by livestock	0, if less than three spot-checks 1, if three to five spot-checks 2, if six or more spot-checks	
B. Modification present but not recorded at spot-checks	**One bank (or channel)**	**Both banks**
Artificial bed material	1	—
Reinforced whole bank	2	3
Reinforced top or bottom of bank	1	2
Resectioned bank	1	2
Embankment	1	1
Set-back embankment	1	1
Two-stage channel	1	3
Weed-cutting	1	—
Bank-mowing	1	1
Culvert	8 for each	
Dam, weir, ford	2 for each	
C. Scores for features in site as a whole	**One**	**Two or more**
Footbridge	0	0
Roadbridge	1	2
Enhancements, such as groynes	1	2
Site partly affected by flow control	1	
Site extensively* affected by flow control	2	
Partly realigned channel**	5	
Extensively* or wholly realigned channel**	10	

* Extensive means at least a third of channel length.
** Information from map.

Appendix A: Habitat modification score sheet (Raven et al., 1998). The quantity of each factor is added up and attributed the value given based on its perceived impact on the river system. Due to recording the quantities it may be less efficient for larger watercourses but for the relatively low order streams being studied it gives a holistic score of direct human impact.



Appendix. B: Non-metric multidimensional Scaling graphs displaying the groups found in cluster analysis by their dissimilarity in n-dimensional space. Due to variance in the data these do not display the dissimilarity as clearly as dendrograms. Panel A describes the biological category (Stress: 0.02821393, rmse: 0.003744588, max residual: 0.009550462, 20 runs), Panel B describes the chemical category (Stress: 0.01918504, rmse: 0.008374715, max residual: 0.008374715, 104 runs), Panel C describes the hydrological category (Stress: 0.02612993, rmse: 0.001480237, max residual: 0.003923277, 20 runs) and Panel D describes the physical category (Stress: 0.03931995, rmse: 0.009025184, max residual: 0.02430246, 20 runs).



Appendix. C: Silhouette widths produced by different quantities of clusters in each category of cluster analysis. Average silhouette width is a measure by which a group can maintain least possible difference between the cases (rivers) that it contains. Therefore, it is a measure of membership to the group of each of these rivers and the optimum number of clusters maximises this.

Appendix. D: Table displaying where rivers are found in the same groups depending on the category of cluster analysis which is being used to sort them. X's in cells dictate that data was not available. Green squares are assigned where pairings of rivers are in the same clusters regardless of category. Categories are displayed by their order of magnitude so that it is possible to understand in which category rivers are together and in which they are not. Physical presence is denoted by 1, Biological by 10, chemical by 100 and hydrological by 1000.

River name	Bailey	Blackflos	Bradford	Coley	Cringle	Dover	Dowles	East Glen	Elmswell	Foston	Great	Hadley	Harpers	Heighing	Henmore	Holywell	Lonco	Lymn	Meece	Mires	North	Oldcotes	Oulton	Pointon	Potford	River	River	River	River	River	River	River	River	River	River Lud	River	River	River Rea	River	River	River	River	River	River	Rothley	Slade	Spen	Stainfiel	West	Willow
Bailey Brook	1111																																																
Blackfloss Beck	1011	1111																																															
Bradford Beck	101	1	1111																																														
Coley Brook	1100	1000	110	1111																																													
Cringle Brook	1111	1011	101	1100	1111																																											
Dover Beck	1011	1111	1	1000	1011	1111																																										
Dowles Brook	1111	1011	101	1100	1111	1011	1111																																									
East Glen River	1x01	1x01	0x11	1x10	1x01	1x01	1x01	1x11																																								
Elmswell Beck	xx11	xx11	xx01	xx00	xx11	xx11	xx11	xx01	xx11																																							
Foston Beck	100	0	110	111	100	0	100	0x00	xx00	1111																																						
Great Eau	111	11	1101	100	111	11	111	0x01	xx11	100	1111																																					
Hadley brook	1111	1011	101	1100	1111	1011	1111	1x01	xx11	100	101	1111																																				
Harpers Brook	1111	1011	101	1100	1111	1011	1111	1x01	xx11	100	101	1111	1111																																			
Heighington Beck	1x10	1x11	0x01	1x00	1x11	1x11	1x11	1x01	xx11	0x00	0x01	1x11	1x11	1x11																																		
Henmore Brook	1111	1011	101	1100	1111	1011	1111	1x01	xx11	100	101	1111	1111	1x11	1111																																	
Holywell Brook	1111	1011	101	1100	1111	1011	1111	1x01	xx11	100	101	1111	1111	1x11	1111	1111																																
Lonco Brook	1111	1011	101	1100	1111	1011	1111	1x01	xx11	100	101	1111	1111	1x11	1111	1111	1101																															
Lymn	1x11	1x11	0x01	1x00	1x11	1x11	1x11	1x01	xx11	0x00	0x01	1x11	1x11	1x11	1x11	1x11	1x11	1x11																														
Meece Brook	101	1	111	110	1001	1	101	0x01	xx01	1110	101	101	101	0x01	101	101	101	0x01	1111																													
Mires Beck	1111	1011	101	1100	1111	1011	1111	1x01	xx11	100	101	1111	1111	1x11	1111	1111	1111	1x11	101	1011																												
North Brook	1011	1111	1	1000	1011	1111	1011	1x01	xx11	0	1	1011	1011	1x11	1011	1011	1011	1x11	1	1011	1111																											
Oldcotes Dyke	11	111	1001	0	11	111	11	0x01	xx11	0	1011	11	11	0x11	11	11	11	0x11	1	11	111	1111																										
Oulton Beck	1101	1001	111	1110	1101	1001	1101	1x01	xx01	110	101	1101	1101	1x01	1101	1101	1101	1x01	111	1101	1001	1	1111																									
Pointon Lode	110	10	100	101	110	10	110	0x00	xx10	1101	110	110	110	0x10	110	110	110	0x10	1100	110	10	10	100	1111																								
Potford Brook	1110	1010	100	1101	1110	1010	1110	1x00	xx10	101	110	1110	1110	1x10	1110	1110	1110	1x10	100	1110	1010	10	1100	101	1111																							
River Bain	1100	1000	110	1111	1100	1000	1100	1x00	xx00	111	100	1100	1100	1x00	1100	1100	1100	1x00	110	1100	1000	0	1110	111	1111	1101																								
River Brant	1xx1	1xx1	0xx1	1xx0	1xx1	1xx1	1xx1	1xx1	xxx1	0xx0	0xx1	1xx1	1xx1	10x1	1xx1	1xx1	1xx1	10x1	0xx1	1xx1	1xx1	0xx1	1xx1	0xx0	1xx0	1xx0	1xx1																							
River Chater	111	11	1101	100	111	11	111	0x01	xx11	100	1111	111	111	0x11	111	111	111	0x11	101	111	11	1011	101	100	100	100	0xx1	1011																						
River Dove	1111	1011	101	1100	1111	1011	1111	1x01	xx11	100	111	1111	1111	1x11	1111	1111	1111	1x11	101	1111	1011	11	1101	100	1100	1100	1xx1	111	1111																					
River Greet	1x00	1x00	0x10	1x11	1x00	1x00	1x00	1x00	xx00	0x11	0x00	1x00	1x00	1x00	1x00	1x00	1x00	1x00	0x10	1x00	1x00	0x00	1x10	0x11	1x11	1x11	1xx0	0x00	1x00	1x11																				
River Gwash	1011	1111	1	1000	1011	1111	1011	1x01	xx11	0	11	1011	1011	1x11	1011	1011	1011	1x11	1	1011	1111	111	1001	0	1000	1000	1xx1	11	1011	1x00	1111																			
River Hamps	111	11	1101	100	111	11	111	0x01	xx11	100	1111	111	111	0x11	111	111	111	0x11	101	111	11	1011	101	100	100	100	0xx1	1111	111	0x00	11	1111																		
River Jordan	1101	1001	111	1110	1101	1001	1101	1x01	xx01	110	101	1101	1101	1x01	1101	1101	1101	1x01	111	1101	1001	1	1111	110	1110	1110	1xx1	101	1101	1x10	1001	101	1111																	
River Leen	101	1	1111	110	101	1	101	0x01	xx01	110	1101	101	101	0x01	101	101	101	0x01	111	101	1	1001	111	110	110	110	0xx1	1101	101	0x10	1	1101	111	1111																
River Lud	1101	1001	111	1110	1101	1001	1101	1x01	xx01	110	101	1101	1101	1x01	1101	1101	1101	1x01	111	1101	1001	1	1111	110	1110	1110	1xx1	101	1101	1x10	1001	101	1111	111	1111															
River Meden	1011	1111	1	1000	1011	1111	1011	1x01	xx11	0	11	1011	1011	1x11	1011	1011	1011	1x11	1	1011	1111	111	1001	0	1000	1000	1xx1	11	1011	1x00	1111	11	1001	1	1001	1111														
River Poulter	0	100	10	11	0	100	0	0x00	xx00	1011	0	0	0	0x00	0	0	0	0x00	1010	0	100	100	10	1011	11	10	0xx0	0	0	0x11	100	0	10	10	10	100	1111													
River Rea	111	11	1101	100	111	11	111	0x01	xx11	100	1111	111	111	0x11	111	111	111	0x11	101	111	11	1011	101	100	100	100	0xx1	1111	111	0x00	11	1111	101	1101	101	11	0	1111												
River Ryburn	111	11	1101	100	111	11	111	0x01	xx11	100	1111	111	111	0x11	111	111	111	0x11	101	111	11	1011	101	100	100	100	0xx1	1111	111	0x00	11	1111	101	1101	101	11	0	1111	1111											
River Ryton	1011	1111	1	1000	1011	1111	1011	1x01	xx11	0	11	1011	1011	1x11	1011	1011	1011	1x11	1	1011	1111	111	1001	0	1000	1000	10x1	11	1011	1x00	1111	1	1001	1	1001	1111	100	11	11	1111										
River Sow	110	10	100	101	110	10	110	0x00	xx10	1101	110	110	110	0x10	110	110	110	0x10	1100	110	10	10	100	1101	101	101	0xx0	110	110	0x01	10	110	100	100	100	10	1001	110	110	10	1111									
River Whitting	1	101	1011	10	1	101	1	0x01	xx01	10	1001	1	1	0x01	1	1	1	0x01	11	1	101	1101	11	10	10	10	0xx1	1001	1	0x10	101	1001	11	1011	11	101	110	1001	1001	101	0	1111								
River Witham	1101	1001	111	1110	1101	1001	1101	1x01	xx01	110	101	1101	1101	1x01	1101	1101	1101	1x01	111	1101	1001	1	1111	110	1110	1110	1xx1	101	1101	1x10	1001	101	1111	111	1111	1001	10	101	101	1001	100	11	1111							
River Worfe	1011	1111	1	1000	1011	1111	1011	1x01	xx11	0	11	1011	1011	1x11	1011	1011	1011	1x11	1	1011	1111	111	1001	0	1000	1000	1xx1	11	1011	1x00	1111	11	1001	1	1001	1111	100	11	11	1111	10	101	1001	1111						
Rothley Brook	111	11	1101	100	111	11	111	0x01	xx11	100	1111	111	111	0x11	111	111	111	0x11	101	111	11	1011	101	100	100	100	0xx1	1111	111	0x00	11	1111	101	1101	101	11	0	1111	1111	1011	110	1001	101	11	1111					
Slade Brook	1000	1100	10	1011	1000	1100	1000	1x00	xx00	11	0	1000	1000	1x00	1000	1000	1000	1x00	10	1000	1100	100	1010	11	1011	1011	1xx0	0	1000	1x11	1100	0	1010	10	1010	1100	111	0	0	1100	1	110	1010	1100	0	1111				
Spen Beck	101	1	1111	110	101	1	101	0x01	xx01	110	1101	101	101	0x01	101	101	101	0x01	111	101	1	1001	111	110	110	110	0xx1	1101	101	1x10	1	1101	111	1111	111	1	10	1101	1101	1	100	1011	111	1	1101	10	1111			
Stainfield Beck	1101	1001	111	1110	1101	1001	1101	1x01	xx01	110	101	1101	1101	1x01	1101	1101	1101	1x01	111	1101	1001	1	1111	110	1110	1110	1xx1	101	1101	1x10	1001	101	1111	111	1111	1001	10	101	101	1001	100	11	1111	1001	101	1010	111	1111		
West Glen River	0x01	0x01	0x11	0x10	0x01	0x01	0x01	0x01	xx01	1x10	0x01	0x01	0x01	0x01	0x01	0x01	0x01	0x01	1x11	0x01	0x01	0x01	0x11	1x10	0x10	0x10	0xx1	0x01	0x01	0x10	0x01	0x01	0x11	0x11	0x11	0x01	1x10	0x01	0x01	0x01	1x00	0x11	0x11	0x01	0x01	0x10	0x11	0x11	1111	
Willow Brook	111	11	1101	100	111	11	111	0x01	xx11	100	1111	111	111	0x11	111	111	111	0x11	101	111	11	1011	101	100	100	100	0xx1	1111	111	0x00	11	1111	101	1101	101	11	0	1111	1111	11	110	1001	101	11	1111	0	1101	101	0x01	1111

Appendix. E: Photos from each field site in the study. Notably: Coley Brook, Foston Beck, Pointon Lode, Potford Brook, River Bain, River Greet, River Poulter, River Sow and Slade Beck are all considered rural streams by the physical classification. Bradford beck, the River Leen and Spen Beck are all found to occur together regardless of the category of analysis and are classified as the urban river group. From these snap shots other streams could arguably be seen to be similar levels of rural or urban on the surface.

Bailey Brook

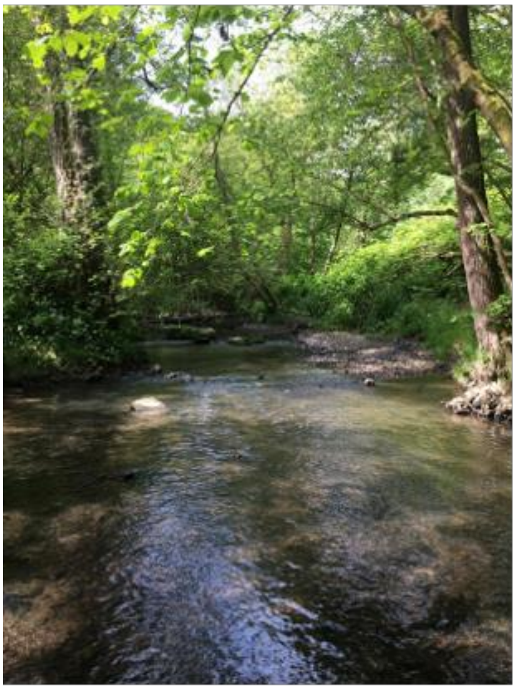
Blackfloss Beck

Bradford Beck

Coley Brook

Cringle Brook

Dover Beck



Dowles Brook

East Glens River

Elmswell Beck

Foston Beck

Great Eau

Hadley Brook

Harpers Brook

Heighington Beck

Henmore Brook

Holywell Beck

Lonco Brook

Lymn

Meece Brook

Mires Beck

North brook

Oldcoates Dyke

Oulton Beck

Pointon Lode

Potford Brook

River Bain

River Brant

River Chater

River Dove

River Greet

River Gwash

River Hamps

River Jordan

River Leen

River Lud

River Meden

River Poulter

River Rea

River Ryburn

River Ryton

River Sow

River Whitting

River Witham

River Worfe

Rothley Brook

Slade Brook

Spen River

Stainfield Beck

West Glens Beck

Willow Brook