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Stream ecosystem responses to restoration across urban tributaries of the River Thames, London

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STREAM ECOSYSTEM RESPONSES TO
RESTORATION ACROSS URBAN
TRIBUTARIES OF THE RIVER THAMES,
LONDON

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1350080

*A thesis submitted to King's College London for the degree of Doctor of
Philosophy*

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December 2018

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ERRATA CORRIGE

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147	7	<i>A. aquaticus</i>	<i>E. testacea</i>

Date

16/09/2022

Signed by





Insert photo: A restored reach of the River Wandle, Carshalton (Author's own)

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Abstract

A common assertion of reach-scale restoration is that modifying in-stream morphology can foster nutrient removal and improve ecological community structure. However, the mechanisms driving structural responses, specifically nutrient processing and decomposition activity at the sediment-water interface are often overlooked in stream assessments. The primary aim of this PhD was to measure the effectiveness of urban river restoration through examination of reach-scale physical, chemical and biological processes which affect biogeochemical process rates and ecological community structure. Restored reaches in this study were generally characterised by in-stream structures including woody debris and gravel material aimed at improving hydromorphological diversity. A novel nitrogen flux assay was developed to simulate nutrient processing across the sediment-water interface in response to physical disturbances and biogeochemical activity. Results revealed equivocal responses to river restoration, a likely legacy effect of urban pollution. Decomposition assays were used to examine ecosystem function using a range of natural substrates (leaf litter, tea bags and cotton strips), although overall rates of breakdown between restored and unrestored sites were negligible. However, the benefits of using standardised, experimental substrates such as cotton strips as surrogates for traditional leaf litter were noted. Fish and macroinvertebrate surveys were used to measure abundance, biomass, richness and community structure. Fish abundance as well as macroinvertebrate biomass and richness were significantly higher at restored sites (i.e. particularly the River Wandle and Hogsmill) and highlighted improvements in ecological community structure. However, restoration failed to modify ecosystem function for either nitrogen processing or organic matter decomposition. These results highlight the need to decipher drivers of ecological recovery through functional stream assessments focused on processes such as primary production.

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Chapter 1: Introduction

1.1 General Introduction

Hydrological, biogeochemical and ecological processes are fundamental for maintaining the structural and functional integrity of aquatic ecosystems. Ecosystem ‘functions’ refer to the interactions between organisms and their habitats (Lecerf and Richardson, 2009). These interactions are driven by ‘mechanisms’ which influence the ‘structure’ of the environment. In urban streams, natural ecosystem structure and functions can be off-set by several factors. These include resource availability such as organic matter for macroinvertebrates which depend on allochthonous (terrestrial vegetation inputs), autochthonous (primary production within the stream) and organic matter transported from upstream (Johnson *et al.*, 1995; Gomi *et al.*, 2002). Organic matter processing, particularly dissolved organic matter (DOM), is fundamental for moderating biogeochemical cycling, however alterations to this resource can have significant effects on carbon processing (Khamis *et al.*, 2017).

Worldwide, population growth and urbanisation have been detrimental to stream networks draining urban land (Paul and Meyer, 2001; Meyer *et al.*, 2005; Walsh *et al.*, 2005; Vietz *et al.*, 2014). The “urban stream syndrome” provides a framework for diagnosing problems associated with urbanisation including physical habitat modifications, hydrological alterations and elevated nutrient loads (Meyer *et al.*, 2005; Walsh *et al.*, 2005). Urban growth increases impervious surface cover (ISC), alters flow regimes and triggers high-magnitude “flashy” events (Walsh *et al.*, 2005; Kaushal *et al.*, 2011; Eloisege and Sabater, 2013). Increasing frequencies of storm events encourage scouring of stream beds and removal of algal biomass, fundamental for stream productivity (Katz *et al.*, 2018). This reduces particle organic matter, inhibiting fundamental ecosystem processes including heterotrophic respiration and nutrient

attenuation (Meyer *et al.*, 2005). Increasingly, homogenisation of stream bed sediments has led to geomorphological and hydrological disconnectivity between the channel and riparian zone, leading to ecological degradation (Walsh *et al.*, 2012; Hawley *et al.*, 2013; McGrane, 2016).

Excess nitrogen loading is a major factor affecting ecosystem functioning. Disruptions to nutrient cycling and food webs can alter ecosystem processes, including leaf litter decomposition, imperative for supporting a range of consumer species (Davis and Koop, 2006; Walsh *et al.*, 2012). Graded responses to nutrient pollution can result from human activity, for example moderate stream nutrient enrichment can accelerate breakdown by stimulating microbial conditioning and invertebrate consumption (Irons *et al.*, 1988). However, beyond threshold concentrations, wastewater discharges can induce anoxia, mobilise heavy metals and smother benthic organisms (Drury *et al.*, 2013). Litter breakdown driven by invertebrates is often considered to be more sensitive to nutrient pollution compared to that of microbes which can elicit a hump-shaped breakdown response rate (Friberg *et al.*, 2011). In a pan-European study examining leaf litter breakdown in 100 streams across a greater than 1000-fold nutrient gradient, reduced leaf litter breakdown was assumed to be driven by strong nutrient limitation (Woodward *et al.*, 2012). This study highlighted the importance of measuring structural indices (such as water chemistry, hydrogeomorphology and biological diversity metrics) as well as functional measures (such as leaf litter breakdown rates, whole-system metabolism and nutrient spiralling) to understand ecosystem health.

The River Thames is the second longest river in the United Kingdom, with a catchment area of 9,948 km², comprising 600 km of rivers and streams. In 2017, it was reported that of the 39 rivers in the Greater London Authority boundary, only one had been classified

as achieving ‘good’ ecological potential under the EU Water Framework Directive (WFD) (Zoological Society of London, 2017).

River restoration projects typically focus on treating the symptoms (e.g. restoring morphology), rather than focussing on the cause of degradation (Vietz *et al.*, 2014). Structural parameters are commonly assessed (e.g. physical habitat, fish and invertebrate community structure) while ecosystem functions (e.g. primary production, nutrient cycling and decomposition) are often overlooked (Tiegs *et al.*, 2007; Young *et al.*, 2008; Tiegs *et al.*, 2013). This is problematic as these measures fail to account for factors which impact upon biodiversity, such as productivity and disturbance regimes (Palmer *et al.*, 2010).

Increasingly, environmental, legislative and social drivers have contributed to restoration informed by hydrological, physicochemical and ecological processes. This has coincided with a push towards river management which promotes working with natural processes (Gilvear, 1999; Clifford, 2007). In order to achieve WFD goals, the Environment Agency has outlined its intent to achieve “good status” for a minimum of 60% of waters by 2021 (House of Commons Library, 2015). In addition, the Habitats Directive and Biodiversity Action Plan (BAP) have also driven improvements in ecological function. An important part of this is promoting ecological integrity – the ability of an ecosystem to support and maintain ecological functions crucial for community diversity (Rempel *et al.*, 2016). Despite this, there is still a dearth of knowledge surrounding the quantification of functional benefits arising from restoration measures. This thesis examines the physical, biogeochemical and ecological responses to restoration across restored and unrestored river reaches of London through the assessment of structural and functional measures. In doing so, it attempts to understand the mechanisms by which different restoration activities elicit ecosystem responses.

1.2 Research Aims, Objectives and Hypotheses

This thesis utilises traditional and novel stream assessment methods to understand structural and functional responses to restoration across urban tributaries of the River Thames, London. The work is presented in an order which explores functional processes (nitrogen, organic matter decomposition) and supporting structural indicators (fish and macroinvertebrate community structure). This provides a platform to understand the mechanisms by which physical, chemical and biological processes may elicit ecological community responses. The five rivers chosen for this project were selected from the UK River Restoration Centre (RRC) derived from the UK National River Restoration Inventory (NRRI) and from previous works by Smith and Chadwick (2014). These varied from 5 to 29 km in length and displayed a range of urban gradients (impervious cover) ranging from 47-69%. The sites also incorporated a range of “best-practice approaches” which varied in age and restoration approach.

The overall research aim is:

To measure the effectiveness of urban river restoration through examination of reach-scale physical, chemical and biological processes which affect biogeochemical process rates and ecological community structure.

More specific research aims, objectives and hypotheses that will be answered are outlined below according to the relevant chapter:

Chapter 3

Research Aim: *To examine whether restoration influences nitrogen processing across the sediment-water interface in urban tributaries of the River Thames, London.*

Objectives:

- 1: To examine ambient nitrogen (nitrate and ammonium) concentrations and further determine whether nitrogen uptake (net movement into sediment) and regeneration (i.e. a measure of release to surface water) varies between restored and unrestored sites.
- 2: To quantify the effect of physical and biogeochemical activity on nitrogen uptake and regeneration.
- 3: To investigate relationships between chlorophyll-*a*, organic matter, sediment size distribution and nitrogen processing.

Hypotheses:

- 1: Restoration would reduce nitrate and ammonium concentrations and facilitate increased nitrogen uptake at restored sites compared to unrestored sites, owing to the presence of in-stream features driving nitrogen assimilation.
- 2: Physical disturbances and biogeochemical activity would increase regeneration at unrestored sites due to sediment nutrient saturation and increase uptake at restored sites due to higher sediment sink capacities.
- 3: Higher concentrations of chlorophyll-*a* and proportions of organic matter would be linked to nitrogen removal due to facilitated assimilative processes, while higher proportions of fine sediment would be linked to nitrogen regeneration due to an increased sediment storage capacity.

Chapter 4

Research Aim: *To compare decomposition rates across restored and unrestored reaches using multiple assay methods.*

Objectives:

1: To determine whether decomposition rates vary between restored and unrestored sites using coarse and fine mesh bags to decipher macroinvertebrate and microbial contributions to breakdown.

2: To evaluate the effectiveness of different assay techniques including leaf litter, cotton strips and tea bags to measure decomposition.

Hypotheses:

1: Decomposition would differ significantly between restored and unrestored sites, owing to processes including physical abrasion, nutrient processing and biological activity.

2: Cellulose cotton strips would provide a preferential tool to measure decomposition over leaf litter or tea bags due to their uniformity and ease of application.

Chapter 5

Research Aim: *To deliver a quantifiable assessment of fish and invertebrate community structure as an indicator of overall stream health.*

Objectives:

1: To assess fish abundance, biomass and richness across restored and unrestored reaches using 3-sweep catch depletions.

2: To examine macroinvertebrate community structure across restored and unrestored reaches using surber sampling techniques.

3: To compare broad physical habitat characteristics between restored and unrestored sites.

Hypotheses:

- 1:** Fish abundance, biomass and richness would increase at restored sites due to improvements in hydromorphological conditions fostering ecological integrity.
- 2:** Macroinvertebrate density, biomass and richness would increase at restored sites due to improvements in water and habitat quality.
- 3:** Physical habitat heterogeneity would increase at restored sites due to improvements in hydrological heterogeneity and habitat diversity.

1.3 Research Methodology, Novelty and Challenges

This research project used empirical data to examine a range of physical, chemical and biological parameters across urban streams in London. Nitrogen-assays, decomposition experiments, electrofishing and macroinvertebrate surveys were used to gain an overall understanding of ecosystem structure and function. The project combined novel experiments (N assays, decomposition assays) and long-established methods (fish and macroinvertebrate surveys) to assess the broader utility of restoration.

There were limitations in the scope and design of this project which should be acknowledged. Examining restored reaches against ‘reference’ streams was not possible given the high degree of urbanisation in the rivers assessed. Whilst the preferred approach would have been to measure restored and unrestored sites against reference and control sites (before-after-control-impact approach), this was not possible given the lack of long-term monitoring data. For this reason, this project adopted a comparative approach to examine differences between several river reaches which represented a range of restoration activities. In the analysis of this work, two-way ANOVAs were used to determine reach-scale differences.

1.4 Thesis Outline

The following chapters will address the aims outlined in Figure 1.1. The thesis will begin with an overview of the supporting literature, describing the history of river management, urban stream syndrome and evolution of urban river restoration through time. Chapter 2 sets out a site description for each river, outlining historic river uses and documenting restoration practices to date. The aims of each restoration approach are described, outlining how they might elicit desired ecosystem responses. A diagram of each reach is displayed.

Chapter 3 employs a novel flux chamber assay approach to examine nitrogen processing across the sediment-water interface in restored and unrestored river reaches. This was a novel technique developed to understand the dynamics of nitrogen processing in response to physical and biogeochemical activity. The purpose of this technique was to mimic behaviours likely to be observed in urban streams subject to flashy events and nitrogen processing over longer timescales.

Chapter 4 evaluates alternative methods for examining decomposition in urban streams using leaf litter, cellulose cotton strips and tea bags. Rates of different assays are explored and compared between restored and unrestored reaches. Findings are assessed against flow data and nitrogen values to attribute breakdown to various factors.

Chapter 5 examines fish and macroinvertebrate community responses across restored and unrestored river reaches. Statistical tools and spatial modelling are used to understand the relationship between species present, flow and favourable habitats.

Chapter 6 summarises the project findings and evaluates overall ecosystem improvements, highlighting recommendation for future research.

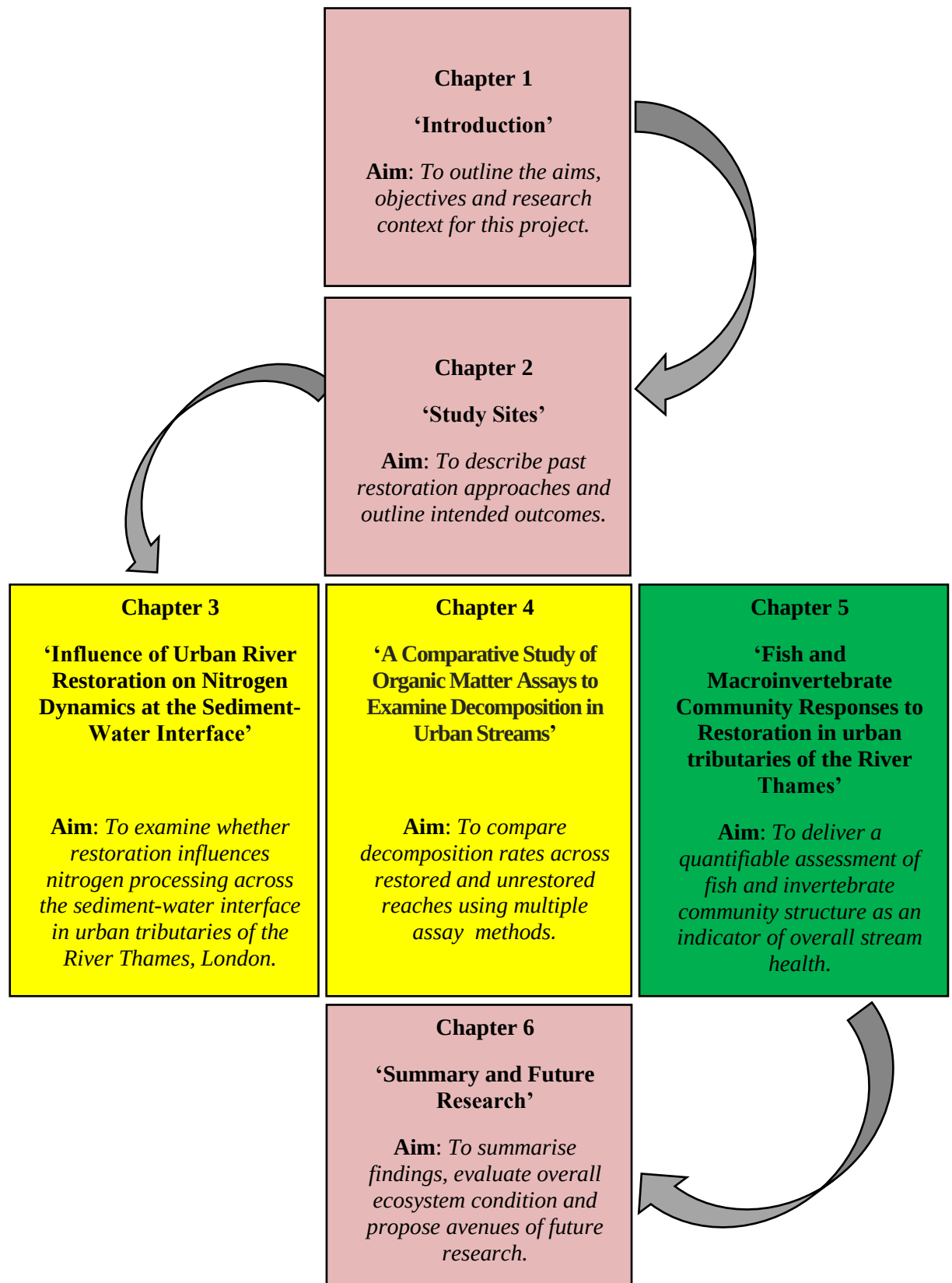


Figure 1.1. Outline of empirical data chapters, showing chapter title, aims and methods. Pink boxes denote introduction, site descriptions and synthesis chapters; yellow boxes denote empirical data chapters examining ecosystem function; green box denotes an empirical data chapter examining ecosystem structure.

1.5 Research Context

This section reviews the key ecosystem processes occurring within urban streams. The first section (1.5.1) outlines historical river management approaches, describes the urban stream syndrome and assesses the effectiveness of urban river restoration. The second section (1.5.2) reviews river nitrogen (N) activity, outlining the importance of N as a key determinant of ecosystem function. This is followed by a section on decomposition activity (1.5.3), reviewing the importance of decomposition processes as a key control on ecosystem activity. Sections 1.5.4 and 1.5.5 review the importance of ecology in urban streams including activity, fish and macroinvertebrates community structure and the mechanisms which influence their composition. The final section (section 1.6) summarises the material covered in the chapter, highlighting the importance of overall ecosystem function.

1.5.1 Urban Rivers

1.5.1.1 Historical River Management

River systems have been described as the ‘lifeblood’ of human civilisation (Harvey and Clifford, 2009) owing to their use for water supply, flow regulation, navigation and flood control (Giller, 2005). The first records of human alterations to rivers in the UK date back to the 1st century A.D., when channels were altered for drainage and transportation (Sheail, 1988). Into the 18th century, several small dams were constructed to support water supply and by the 19th century, increases in river pollution triggered widespread control of rivers characterised by the building of dams and large inter-basin transfers (Sheail, 1988). The first major modifications to rivers were associated with hard engineering approaches focused on controlling river systems and harnessing their resources. This involved implementing straight trapezoidal channels, impoundments and embankments to reduce the frequency and duration of flooding (Kondolf *et al.*, 2006). Up until the middle of the 20th century, rivers were viewed as conduits with which to manipulate the

transport of water, sediment and waste through uniform and ‘stable’ channels (Giller, 2005). Globally, an increasing demand for water supply triggered a global network of rivers that were impounded, straightened and polluted (Downs and Gregory, 2004). A combination of encroaching pressures resulted in rivers which were severely degraded (Giller, 2005).

The environmental movement of the 1960s challenged river managers to consider more environmentally sensitive approaches to restoration (Ormerod, 2004). The introduction of the EU Water Framework Directive (WFD) 2000/60/EC, in combination with increased public awareness of ecosystem services which could be provided by river systems, encouraged scientists and managers to examine river processing from a catchment perspective.

River management approaches have shifted towards understanding water-sediment dynamics and channel connectivity. Fluvial landforms and substrates define habitats for biota and are also sensitive to environmental changes (Clarke *et al.*, 2003). Disrupted connectivity can influence sediment budgets, leading to a reduction in oxygenation of spawning grounds (Gilvear, 1999).

‘Eco-engineering’, the process of using collaborative methods to improve the ecological condition of rivers, is increasingly being used to characterise stream ecosystems (Gilvear, 1999; Figure 1.2). Several studies have reported upon the benefits associated with eco-engineering, especially for fish species benefitting from undercut banks. On the Evan Water, Scotland, salmon populations have stabilised in response to morphological variability geared towards improving channel variability (Gilvear, 1999).

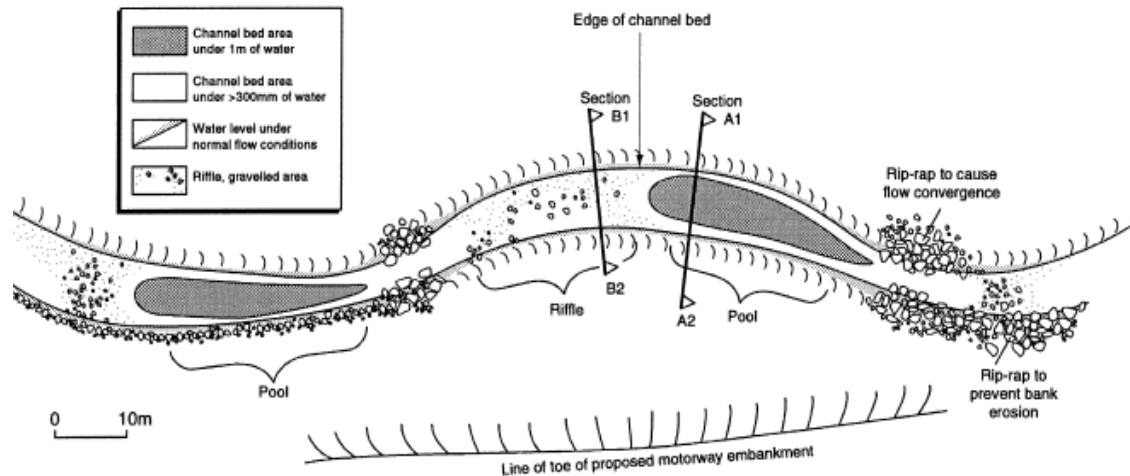


Figure 1.2. Water environmentally sensitive engineering on the Evan Water, Scotland, displaying a meandering design and in-stream structures to encourage depth, flow and sediment variability and designed to improve salmon habitats (Gilvear, 1999).

1.5.1.2 Urban Stream Syndrome

The “urban stream syndrome” is a framework used to evaluate change associated with physical, hydrological and chemical degradation in urban streams (Walsh *et al.*, 2005; Figure 1.3.). Impervious surface cover (partial or complete) is often considered to be a primary cause of degradation and can off-set geomorphological and hydrological connectivity between the stream-channel, hyporheic and riparian zone. Schueler *et al.* (2009) found that stream impairment occurred at impervious surface cover values ranging from as little as 2 to 15%. Geomorphological alterations including over-widening and incising of urban channels can trigger complex sediment-supply problems in urban catchments, reduce riparian vegetation and decrease habitat complexity (Walsh *et al.*, 2005). Pizzuto *et al.* (2000) found that urban watercourses were wider, had shallower pools, lower sinuosity and an overall lower channel roughness compared to rural watercourses. Furthermore, sediment supply to urban channels from channel erosion and hillslope processes was thought to be significant. Infiltration and altered flow regimes including dramatic ascending and descending hydrographs, discharge regimes and alterations to in-stream kinetic power are common impacts of this (Burns *et al.*, 2012).

Episodic flood events in urban streams also alter stream sediment yields which are exacerbated by the removal of vegetation, resulting in reduced vertical, lateral and longitudinal connectivity. Exposed soils can increase catchment sediment yields which are expelled during episodic flood events (Elosegi and Sabater, 2013). This results in lower leaf litter and coarse woody debris inputs, both fundamental attributes of ecological function (Paul and Meyer, 2001). An increase in wastewater and stormwater inputs reduces sediment and water quality due to increases in suspended solids, nutrients, metals, hydrocarbons, bacteria and lowered dissolved oxygen.

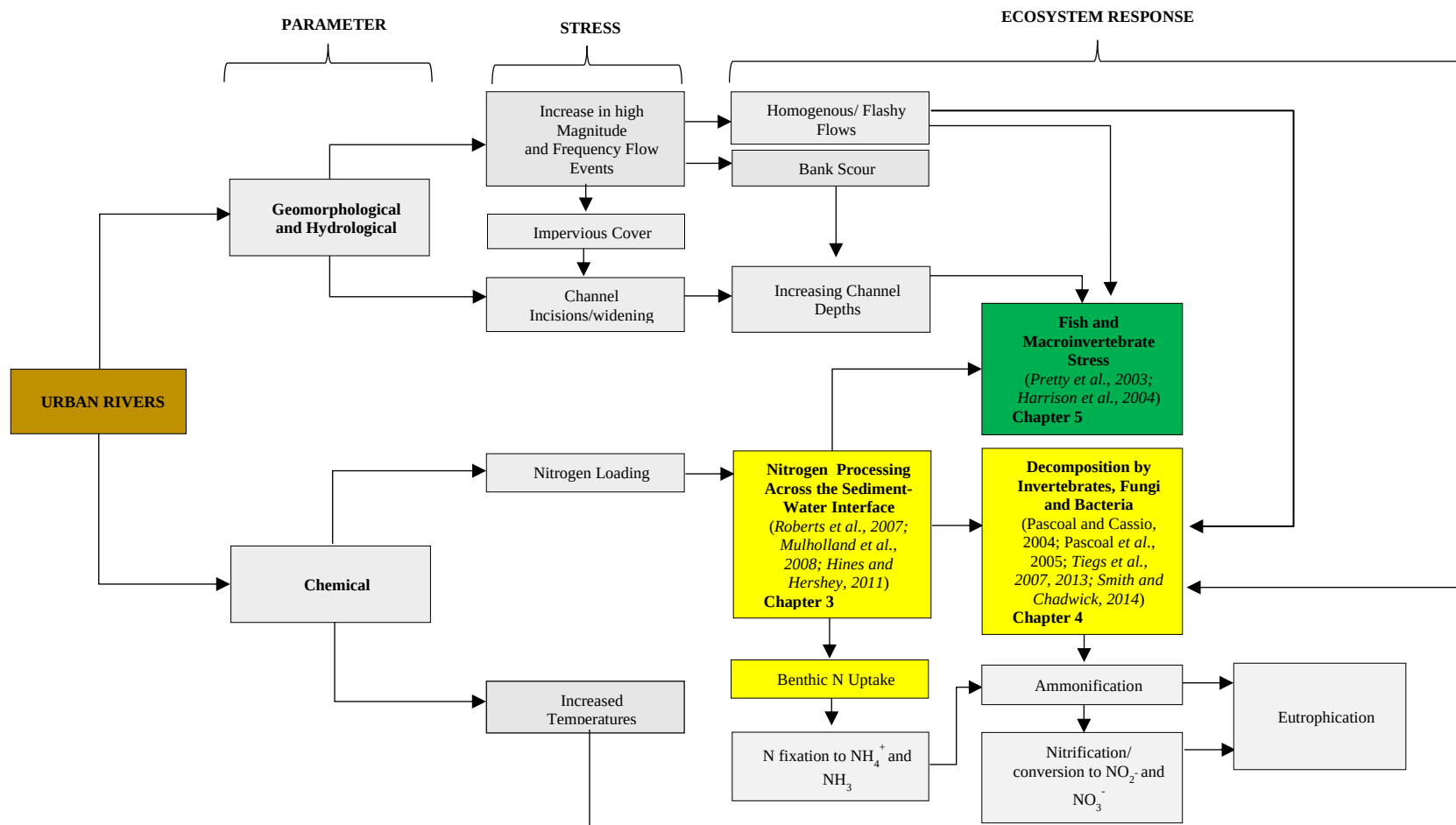


Figure 1.3. Conceptual model highlighting characteristics, stressors and ecosystem responses to urbanisation in aquatic ecosystems applicable to this project. Yellow boxes denote empirical data chapters examining ecosystem function. Green boxes denote empirical data chapters examining ecosystem structure. Yellow and green boxes both include relevant authors.

1.5.1.3 Urban River Restoration

Worldwide, environmental, legislative and social factors are the main drivers of river restoration projects. The EU Water Framework Directive (2000/60/EC) commits European Union member countries to achieving good qualitative and quantitative status of all water bodies. This is based on several biological (fish, macroinvertebrates, aquatic flora), hydromorphological (bank structure, river continuity, substrate) and physico-chemical (temperature, oxygen, nutrient) factors, in addition to chemical quality (environmental quality standards for river basin pollutants) conditions. In addition to this, increased awareness of ecosystem services and natural capital gains that can be provided from restoration projects including climate regulation and water supply have also driven river management schemes.

‘Full restoration’, the notion of returning a river to its original state, is often seen as an idealistic goal in catchments with a legacy of infrastructural developments and pollution (Downs and Gregory, 2004). It is often difficult, if not impossible to determine pre-restoration conditions or quantify improvements, particularly on a catchment scale (Mainstone and Holmes, 2010). More commonly, river restoration is classified into more discreet sub-terms including: creation - developing a new resource or structure; enhancement - delivering any measurable improvement in environmental quality; rehabilitation - delivering partial improvements in river structure or function; or restoration - returning to a pre-restoration state (Figure 1.4).

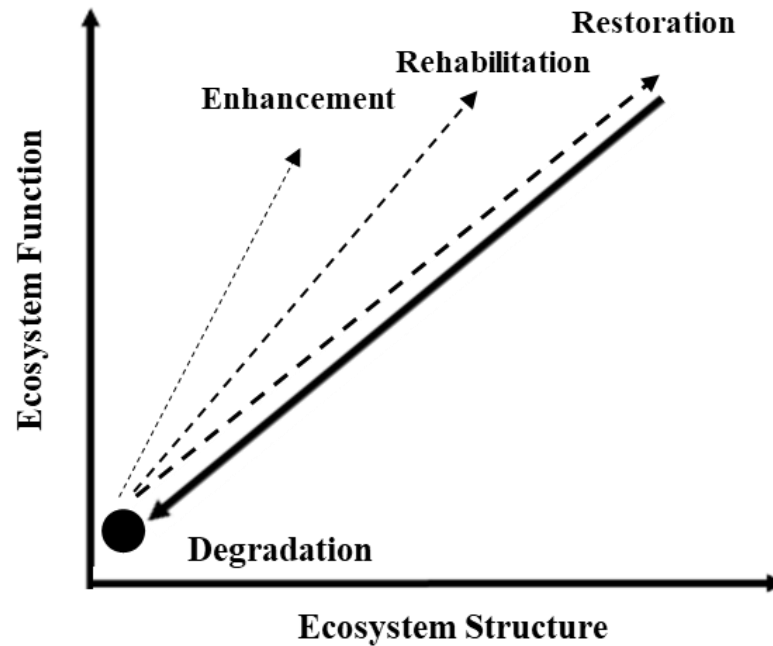


Figure 1.4. Illustration of different levels of restoration: enhancement, rehabilitation and restoration. Movements from degraded to enhanced, rehabilitation and restoration are symbolised by lines in order of condition (degraded to restored). Structural parameters include species diversity and functional parameters include nutrient dynamics and decomposition (Adapted from Downs & Gregory, 2004; Findlay & Taylor, 2006).

Changing societal attitudes towards nature have shaped river restoration practices throughout time (Clifford, 2007). Following the movement towards more ecologically-sensitive engineering methods in the 1980s and 1990s, most projects sought to improve flow regime and natural patterns of connectivity. Bunn and Arthington (2002) realised the importance of flow diversity at a patch, reach and catchment scale in supporting different species distributions. Features such as woody riparian vegetation can support geomorphological diversity and mediate sediment budgets in low energy streams (Gurnell *et al.*, 2014). More recent studies have emphasised the value of flood and process-based management for supporting river restoration (Grabowski *et al.*, 2014).

Restoration has been linked to nitrogen (N) uptake in urban streams (Roberts *et al.*, 2007; Hines and Hershey, 2011; Figure 1.5). Janes *et al.* (2017) reported higher removal efficiencies and improvements in water quality in Johnson Creek, Portland, USA, linked to the establishments of swales, lower outflow velocities, deposition and uptake by plants.

Further benefits associated with this include the potential for large woody debris (LWD) to attenuate nitrate (NO_3^-) through increased streambed residence times and hyporheic exchange (Shelley *et al.*, 2017).

Flow conditions are an important consideration of river restoration projects given their role in influencing physical and biotic habitats (Hill *et al.*, 1991; Bunn and Arthington, 2002; Figure 1.5). Modern restoration approaches account for the complexities and dynamisms associated with natural fluvial systems (Newson and Large, 2006; Figure 1.6). Flow determines physical habitats and biotic structure as well as longitudinal and lateral connectivity. Whilst lateral connectivity between river channels, floodplains and groundwater can encourage structural heterogeneity and habitat biodiversity, lateral expansions of floodplain habitats during flood events can provide suitable spawning and nursery habitats for fish species (Ward *et al.*, 1999). Flow regimes are critical for supporting key life-cycle events of spawning and reproduction. Importantly, modifying flow at different spatial scales including patch (cobbles), reach (riffle pools) and catchment (channel) can support an array of different aquatic species (Bunn and Arthington, 2002). Several studies have noted significant improvements in hydrological diversity resulting from the implementation of in-stream structures across restored reaches (Pretty *et al.*, 2003; Haase *et al.*, 2013; Leps *et al.*, 2016).

Ecological responses to hydromorphological-based restoration projects have been equivocal (Brown, 2003; Bernhardt and Palmer, 2011; Haase *et al.*, 2013). Harrison *et al.* (2004) found that engineered riffle-pools and flow deflectors led to a small degree of improvement in invertebrate taxa not observed in other habitats. Marginal sites and in-stream macrophytes were higher in number, abundance and richness.

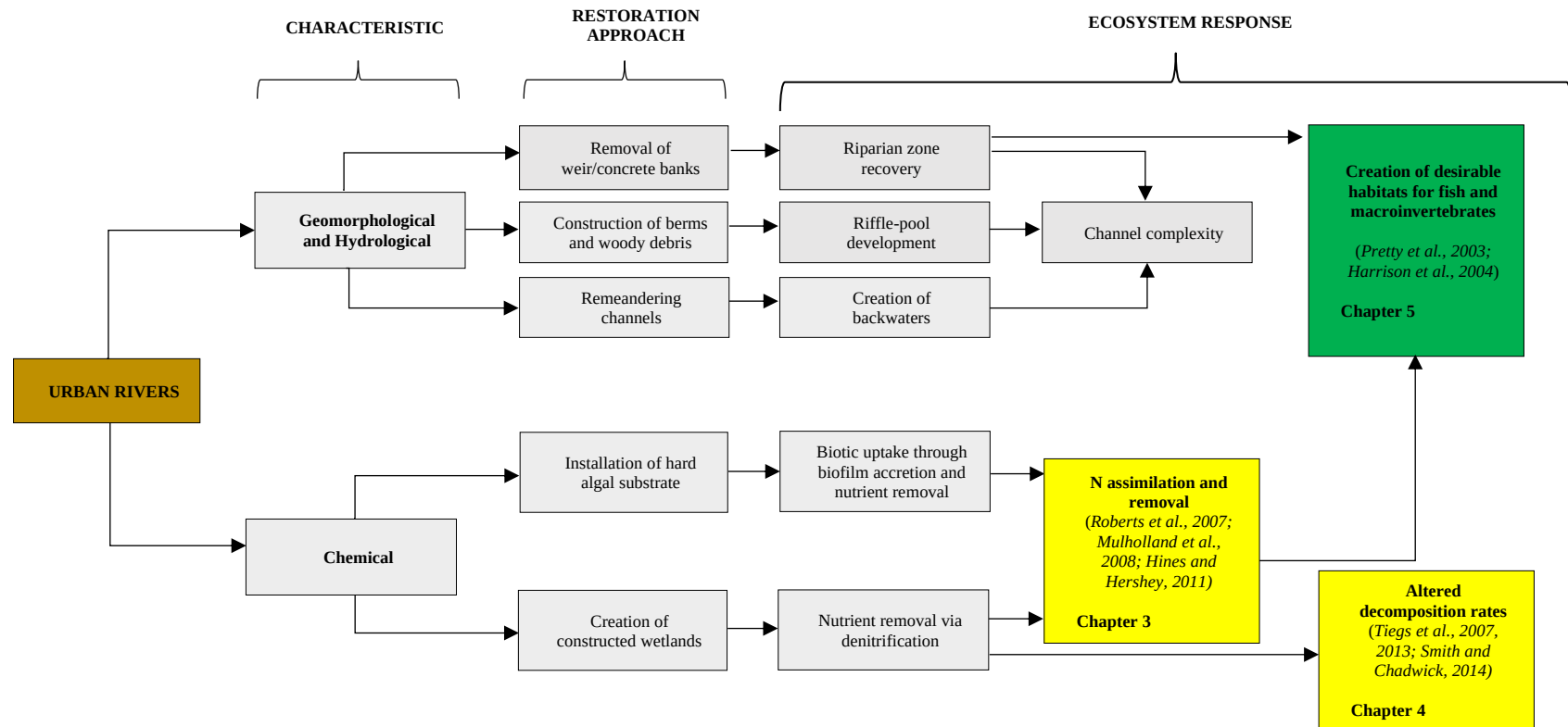


Figure 1.5. Conceptual model highlighting characteristics, stressors and ecosystem responses to restoration in aquatic ecosystems. Yellow boxes denote empirical data chapters examining ecosystem function. Green boxes denote empirical data chapters examining ecosystem structure. Yellow and green boxes both include key relevant authors.

Habitats based on the morphological characteristics of aquatic plants

Trailing vegetation: Feeding, mating and oviposition zone for mature stages of otherwise aquatic plants

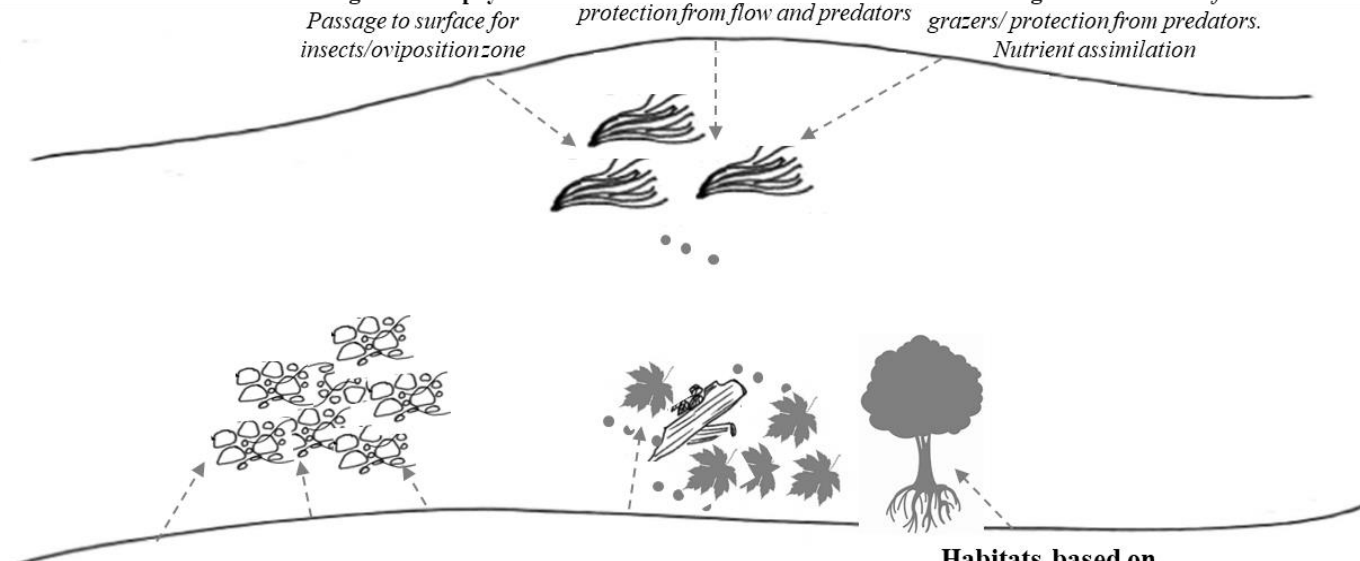
Emergent macrophytes: Passage to surface for insects/oviposition zone

Floating-based plants: Passage to surface for emerging insects

Submerged, broad/fine leaved: Provides surfaces for attachment/ protection from flow and predators

Mosses: Oviposition in faster flows/accumulation of fine sediment and organic matter for physical substrate and food; Nutrient assimilation

Macroalgae: Food source for grazers/ protection from predators. Nutrient assimilation



Substrate

Exposed rocks & boulders: Hygropetric zone – a thin film of water of rock surface providing habitat for macroinvertebrates

Sand: Supports small, specialised macroinvertebrates. Accumulates around features such as woody debris, enhancing biological activity

Cobbles: hyporheic zone with interstitial spaces. Fish spawning sites/ supports macroinvertebrates

Silt: Provides a detrital based habitat for macroinvertebrates.

Habitats based on riparian/terrestrial vegetation

Leaf-litter: Food for shredders; site for production and aggregation of fine particle organic matter and subsequent capture of nutrients

Woody debris: Direct use by borers; increases channel stability

Tree roots: Potentially important sites for specialised species

Figure 1.6. Physical biotopes and functional habitats (Adapted from Newson and Newson, 2000).

However, other studies (Palmer *et al.*, 2010) have noted that there is little evidence to suggest that habitat heterogeneity influences stream invertebrate diversity. Fish assemblages, abundance, richness and diversity have increased among some restored sites (Pretty *et al.*, 2003). In other studies measuring ecological function between restored and unrestored sites, macroinvertebrate community structure has shown no improvement between restored and unrestored reaches (Lepori *et al.*, 2005; Smith and Chadwick, 2014). To date, it has been suggested that river restoration practices have focused too heavily on structural indicators rather than associated processes and functions (Woodward *et al.*, 2012).

Evaluating restoration success is important for understanding reasons for failure and informing avenues of future work (Woolsey *et al.*, 2007). In order to maximise the chances of success of river restoration projects, it is important to set out clear pre and post restoration goals to determine geomorphological, biological and societal baselines and gains, both for structural and functional indices which take account of seasonal and hydrological variations.

1.5.2 Nitrogen Processing in Urban Rivers

Nitrogen (N) is critical for regulating the productivity of freshwater ecosystems for a range of primary producers, including aquatic plants and algae. Urban watersheds receive N inputs from atmospheric deposition, point sources including waste water treatment plants (WWTPs) and combined sewer outflows (CSOs); diffuse land-based sources (e.g. manure and fertilisers); unregulated discharges; leaky septic pipes and misconnections (Simon *et al.*, 2010; Kaushal and Belt, 2012) (Figure 1.7). The rate at which N enters the aquatic environment depends upon several factors including: the mode of transport (e.g. runoff, interflow, subsurface flows), land use, geology, soils and drainage infrastructure (Lintern *et al.*, 2018). Urban rivers receive efficient inputs of N due to rapid transport through runoff from impervious surfaces and artificial drainage networks. During high rainfall spells, impervious surface cover, sparse vegetation and soil of low permeability limit potential for biological removal of N (Wollheim *et al.*, 2005). N is also transported through interflow and through groundwater. This is particularly common in catchments subject to altered hydrological and biogeochemical regimes (Lintern *et al.*, 2018).

Various biotic reactions including N fixation, ammonification, denitrification, oxidation, nitrification, reduction and inorganic N assimilation into microbial biomass regulate the bioavailability of N. Nitrate is the most dominant source of N in aquatic environments due to its high solubility and leaching capacity (Craig *et al.*, 2008). Ammonium (NH_4^+) is less common than NO_3^- since it is readily immobilised, adsorbing to negatively charged clay particles and organic matter (OM) and is often nitrified in small streams (Figure 1.7). Organic N can also present in the form of dissolved or particulate organic N (Newcomer Johnson *et al.*, 2016). Permanent NO_3^- removal in streams occurs when NO_3^- is reduced to N_2 or N_2O under anaerobic conditions. In-stream features, including debris dams are often hotspots for removal given their ability to oxidise organic matter (Groffman *et al.*

2005). Removal can also occur at hyporheic and backwater zones through temporary physical and biological processes (Li *et al.*, 2017). In addition, microbes, algae and vegetation can assimilate N after which it can then be released through decomposition and remineralisation.

Nitrogen transfers (i.e. uptake and regeneration) across the sediment-water interface via physical and biogeochemical processes are influenced by a wide range of factors including river discharge, sediment type, water quality and stream metabolism (Maksymowska-Brossard, 2001; Grimm *et al.*, 2005). Changes in flow regime, particularly changes from lotic to lentic water can lead to a build up of nutrients and degrade water quality. The river bed acts as a sink for nitrogen, which can trigger biogeochemical transformations between the sediment and water column (Clavero *et al.*, 2000). In anaerobic conditions, NO_3^- is reduced to NH_4^+ which can be released back into overlying waters following physical disturbances or biogeochemical activity (Roberts *et al.*, 2007; Mulholland *et al.*, 2008; Hines and Hershey, 2011). NH_4^+ and NO_3^- cycling strongly influence stream productivity on both a daily and seasonal basis; if sediments are saturated with N, disturbances can accelerate the release of NH_4^+ and NO_3^- , which in high concentrations is toxic to fish and invertebrate fauna (Randall and Tsui, 2002). While denitrification can lower the effects of N transfers downstream, oversaturation can conversely limit primary production and trigger eutrophication (Grimm *et al.*, 2005).

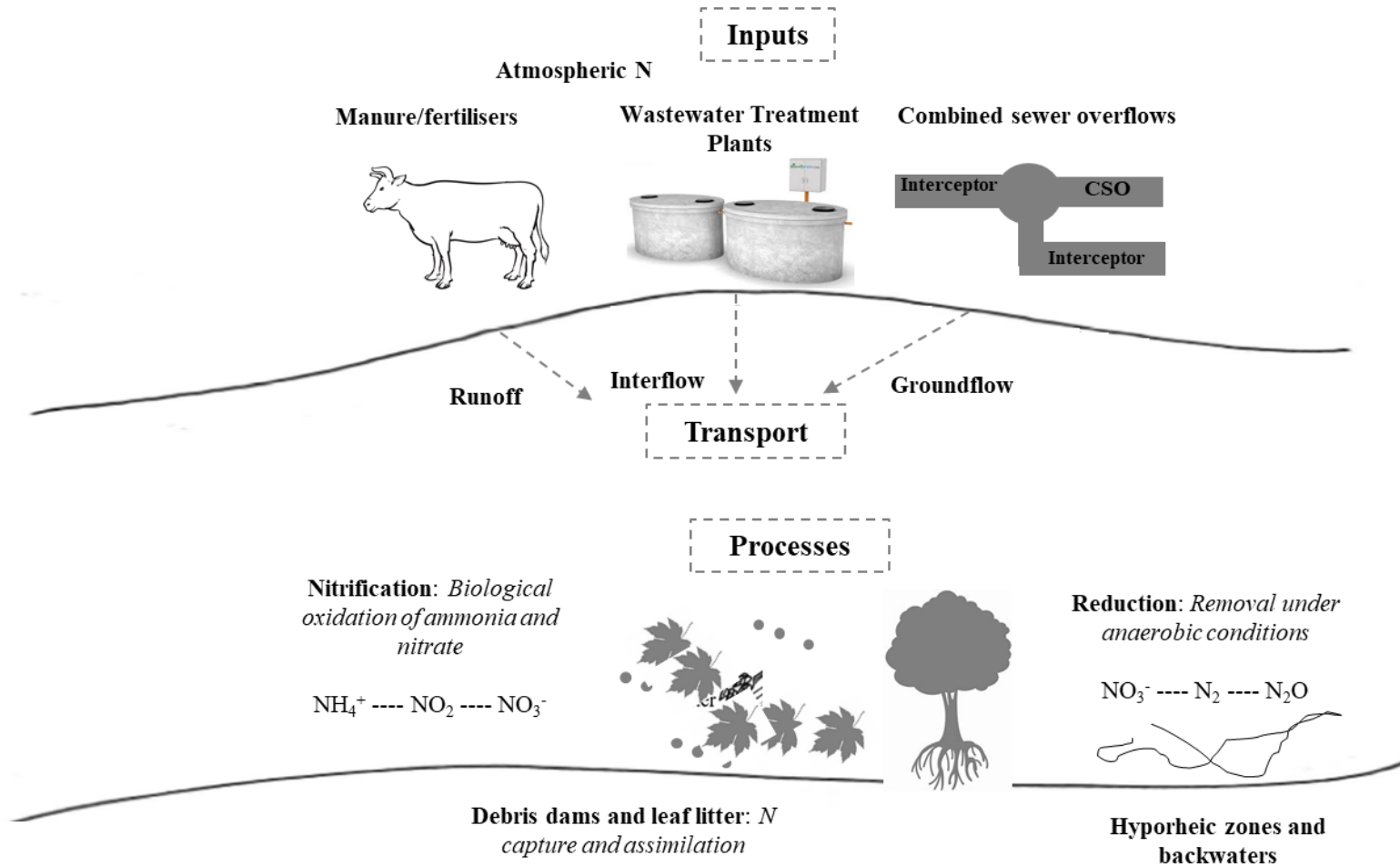


Figure 1.7. Nitrogen processing in urban streams, highlighting the key inputs, transport and processes occurring.

Some studies have pointed to the lack of concrete guidelines for improving the retention and removal of nitrogen in rivers (Craig *et al.*, 2008). Different restoration techniques can elicit various pollutant removal mechanisms (Ren *et al.*, 2015). Proposed methods for reducing N in streams include increasing in-stream carbon, riparian planting, contact between water and benthos and instrumenting stream/terrestrial zone connectivity (Craig *et al.*, 2008) (Figure 1.8). Flow alterations are often prioritised in restoration schemes due to their potential for influencing nutrient removal. It has been proposed that 1-3 order streams exhibit the highest potential for N removal during low and moderate flows and that 50% of N in rivers have the potential to be removed prior to entering coastal environments (Galloway *et al.*, 2004).

Physically modifying channels to encourage increased surface to volume ratios, hydraulic retention and water-benthos contact can trigger N removal in streams (Kasahara and Hill, 2006). Common features include the introduction of woody debris and riffle pools which promote groundwater-surface water mixing and contact with benthos (Seitzinger *et al.*, 2002). In a study of Canadian streams, Kasahara and Hill (2006) examined flow in the hyporheic zone in response to restoration. They found that riffle pool and step constructions encouraged mixing between the surface and groundwater which promoted the removal of N from the channel.

Nitrogen removal by ecological purification has the potential to improve water quality through the implementation of constructed wetlands, parallel coupling multiple-stage wastewater purification, shallow aquifer infiltration and ecological water quality improvements (Ren *et al.*, 2015). In this study, temporal factors including seasonality were found to have a large influence on N removal. For example, N removal was much higher in the summer (2.35 mg l^{-1} in summer 2011, compared to 9.94 mg l^{-1} in winter 2011), a likely result of greater numbers of aquatic plants which encouraged microbial

processes and N removal. In contrast, N removal was lower during colder months, likely due to anaerobic conditions resulting from ice cover which greatly reduced biodegradation. N concentrations were significantly reduced during years where vegetation was well established, highlighting the importance of transition periods to allow for the necessary succession of plants to remove N. Among all cases, volatilisation, nitrification and plant uptake are likely to have been instrumental in N uptake (Ren *et al.*, 2015)

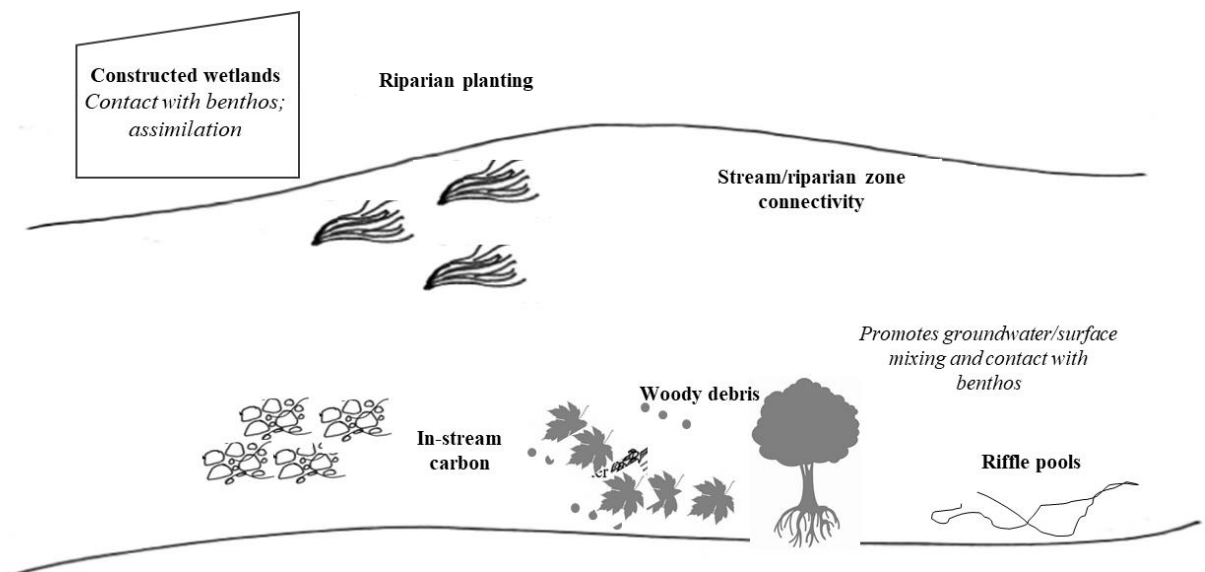


Figure 1.8. Restoration approaches suitable for nitrogen removal in urban streams.

Reconnecting streams with adjacent environments can increase the potential for carbon-rich soils to become saturated with N and reduce loading downstream. Fischenich and Morrow (2000) found that two-stage channels increased contact with riparian vegetation and helped to contain storm flows within the channel. Off-channel management, including constructing connections with wetlands, can facilitate water N removal. Several authors have highlighted the importance of implementing in-stream carbon sources to increase nitrate and ammonium removal from streams (Hall and Tank, 2003; Webster *et al.*, 2003). Increasing carbon in-stream helps to trap fine sediments and nutrients, creating hotspots for denitrification. This provides a source of energy for denitrifying bacteria, which helps to promote anoxia and heterotrophic respiration.

1.5.3 Decomposition in River Ecosystems

Organic matter is an important source of energy in streams for macroinvertebrates and microbes (Figure 1.9). Energy inputs from allochthonous (terrestrial vegetation inputs), autochthonous (primary production within the stream) and organic matter sources, contain humic and fulvic acids, leaf detritus, chlorophyll and microorganisms (Johnson *et al.*, 1995). The physical environment largely controls the potential for macroinvertebrates to effectively utilise different food types (Ward and Stanford, 1983). In headwater streams, riparian vegetation (allochthonous) material is the primary energy source due to limited light availability, which results in a lower photosynthesis: respiration ratio ($P/R < 1$) and dominance of shredders which utilise coarse particle organic matter (CPOM > 1 mm) (Johnson *et al.*, 1995). Mid-order reaches are characterised by greater physical and hydrological diversity, which in turn drive biological diversity (Johnson *et al.*, 1995). Elevated sunlight exposure can increase photosynthesis, while the distribution of fine particle organic matter (FPOM, 50 μm -1 mm) is favoured by collectors which are adapted to transport fine particles. Grazers, characterised by their shearing mechanism, also thrive in these environments (Figure 1.9). Studies of small forested streams have highlighted that these systems are often heterotrophic and dependent upon inputs of CPOM (Vannote *et al.*, 1980). However, even if streams receive large inputs of CPOM, stream retention is a key determinant influencing benthic build-up of CPOM. Retained detritus is broken down by biotic or abiotic factors prior to being transported downstream (Webster and Benfield, 1986). Mechanical fragmentation can off-set in-stream retention and reduce available organic matter.

Reductions or loss of available organic matter through channelisation and increased impervious surface cover can impair important functions carried out by leaf-shredding macroinvertebrates. Shredders are often limited by food availability and human activity associated with reductions in leaf litter inputs can reduce retention. This reduces shredding which is important for supporting other macroinvertebrate species.

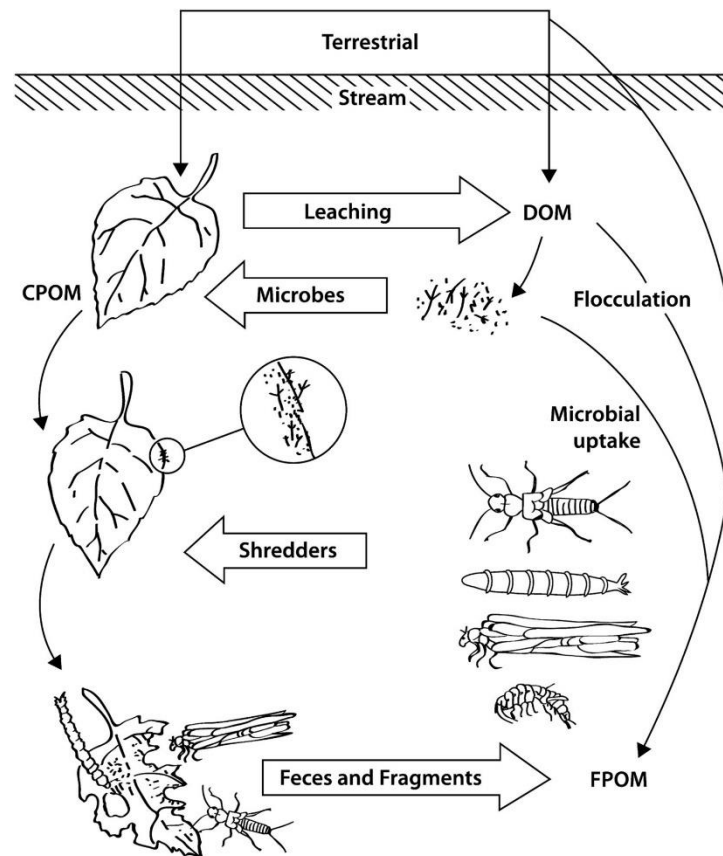


Figure 1.9. Illustration of how alterations in stream mechanisms influence habitat (Ecology Center, 2019).

The greatest amount of biogeochemical activity in river systems occurs at the river bed. This is due to high concentrations of microorganisms in sediments and increased resident durations compared to overlying waters (Tank *et al.*, 2010; Trimmer *et al.*, 2012). Areas of low flow and high macrophyte density trap sediment containing organic matter (dissolved organic carbon and fine particulate organic carbon) which is critical for

supporting oxygen consumption and acceptance of electrons. Furthermore, these allochthonous deposits provide a food source for fish and invertebrates.

Pollution can stimulate leaf litter breakdown. Pascoal and Cassio (2004) determined the leaf litter decomposition of *Alnus glutinosa* in order to understand the effects of pollution on the ecology of the Ave River (North-west Portugal). Elevated leaf litter breakdown was linked to nutrient enrichment. Increased pollution levels also correlated with increases in species density, decreases in macroinvertebrate richness and reductions in the presence of aquatic hyphomycetes. As seen in other studies (Paul and Meyer, 2001; Pascoal *et al.*, 2005), elevated levels of organic and inorganic nutrients have coincided with decreases in macroinvertebrate richness. Understanding the mechanistic relationships between aquatic organisms and litter decomposition rates can provide useful indicators of ecosystem health (Smith and Chadwick, 2014). Rosemond *et al.* (2015) examined the relationship between nutrient input and accelerated terrestrial carbon loss from stream ecosystems. To understand the impact of nutrient enrichment on organic carbon (C) (an important driver of food web dynamics), stream nitrogen (N) and phosphorus (P) were examined over a 27-year period. Results detected a 50% reduction in terrestrial organic C triggered by nutrient pollution when compared to reference conditions. Further analyses found that organic C was depleted via the release of detrital food webs from N and P co-limitation. Rosemond *et al.* (2015) suggest that terrestrial C loss can exceed algal C gains because of nutrient enrichment across river networks.

Lepori *et al.* (2005) examined the effects of restoration on stream ecosystem function by comparing CPOM breakdown and retention between restored, channelised and reference sites. The approach taken involved using artificial alder (*Alnus* spp.) leaves in coarse mesh bags. They sought to understand possible causes of breakdown, accounting for the role of physical and biological activity. Results revealed that CPOM retention did not

differ significantly between restored and unrestored sites. CPOM mass loss was not linked to nutrient conditions, pH, shredder biomass and velocity. However, further investigation highlighted that discharge negatively influenced retention as a result of increased velocities in response to channelisation.

1.5.4 Fish Dynamics in Freshwater Systems

Approximately 45% of all fish species worldwide are dependent on freshwater habitats. Fish are useful indicators of ecosystem health as they highlight sensitivity to pollution and habitat fragmentation (Walsh *et al.*, 2005). Fish species depend upon refuge, feeding and spawning habitats for survival (Northcote, 1984) (Figure 1.10). In functional river ecosystems, fish habitats exhibit a diverse range of environmental conditions which support biological activity and dispersal activity. Refuge and feeding habitats are vital for supporting the survival of freshwater fish and they depend on a diverse range of habitat conditions including flow, depth and turbulence velocities. Hydrological connectivity allows for the free movement of species to migrate up and downstream. This is particularly important for species which depend on different conditions for spawning and for providing refuge from predation (Figure 1.10).

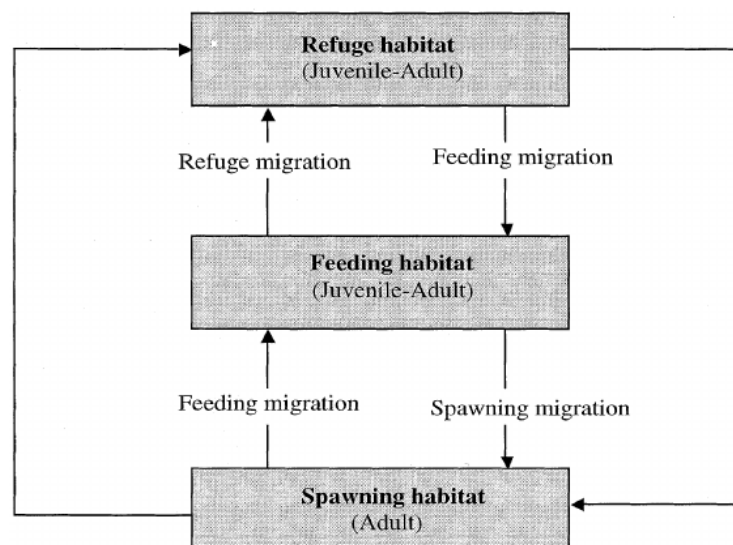


Figure 1.10. Generalised patterns of migration between the three basic habitats utilised by many migratory fish (from Brittain and Braband, 2001, as simplified after Northcote, 1978).

Human activity has significantly altered longitudinal and lateral conductivity (e.g. impoundments), natural flow patterns and habitat structure within urban streams. In-stream structures create a physical barrier to physical, hydraulic (Vowles and Kemp, 2012), chemical (e.g. acid sulphate soil discharge, zones of pollution, low dissolved oxygen) and behavioural processes (e.g. light conditions may induce avoidance behaviour (Vowles and Kemp, 2012)). Historically, there have been an estimated 800,000 low-head weir structures (< 5 m) worldwide (McCully, 1996; Petts, 1984), which have jeopardised feeding, spawning and colonisation habitats (Branco *et al.*, 2012), restricted the movement of migrating species (especially those dependent on migration for spawning) and created areas of geographic isolation, in turn increasing the risk of predation.

Channel impoundments can significantly alter hydraulic conditions upon which fish species depend for swimming and leaping capacities (Figure 1.11). Flow velocity, depth, aeration and turbulence all influence survival potential in addition to important physico-chemical factors such as oxygen and water temperature. Certain fusiform fish such as brown trout *Salmo trutta* and roach *Rutilus rutilus* have been found to have a greater swimming capacity when compared to bullhead *Cottus perifretum* and stone loach *Barbatula barbatula* (Tudorache *et al.*, 2008). In contrasting cases, migration capacity has been linked to swim speed and oxygen consumption.

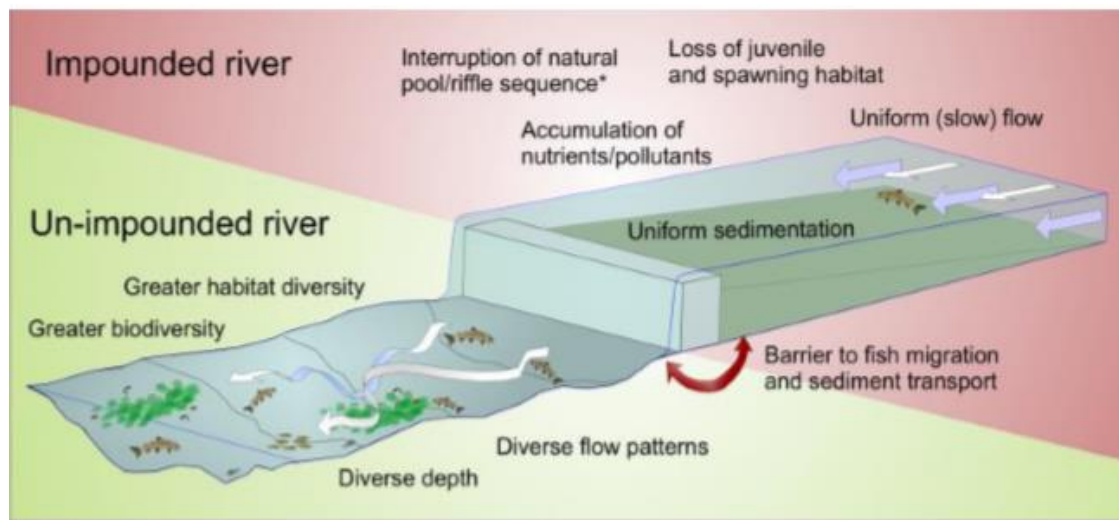


Figure 1.11. Impacts of impounded and un-impounded rivers on channel processes (Wild Trout Trust, 2015).

Sedimentation associated with urban degradation can have harmful effects on fish, both directly (through organ damage) and indirectly (via changes in water quality) (Kemp *et al.*, 2011). Alterations in water velocity can cause damage through erosion of the mucus coating, exacerbated by angular clasts associated with finer sediment sizes. Furthermore, sediment can bind to the gill epithelium, inhibiting gaseous exchange (Kemp *et al.*, 2011). Reduced oxygen levels resulting from fine sediment in the aquatic environment, particularly under low flows and increased temperatures, can lead to an increase in tolerant fish species. Additionally, embryonic eggs (especially salmonids) requiring gravel nests are threatened by reductions in oxygen supply. In urban watersheds, wastewater treatment plants have been found to influence the size, activity and composition of microbial communities (Drury *et al.*, 2013). Areas experiencing high effluent loads were almost indistinguishable from those upstream, showing elevated levels of inorganic nutrients, reductions in population size and a reduced diversity of sediment-based bacterial communities.

Connectivity, referring to the functional exchange of matter, energy and organisms which acts regionally and globally has been an important goal for encouraging habitat connectivity in UK rivers under the WFD. Reconnecting reaches which are disconnected

longitudinally, laterally, vertical and temporally are important for feeding, spawning, nursery habitats and refugia. Removal of fish passes or impoundments which inhibit fish movement, flow regime, sediment transport and nutrient dynamics is recognised as an effective method for improving the condition of fragmented river habitats (Kemp and O'Hanley, 2010).

Increasingly, restoration has been geared towards reducing fine sediment through approaches such as gravel augmentation, implementing in-stream structures (e.g. woody debris, boulders) and gravel cleansing. Gravel augmentation involves replenishing depleted or degraded sediment to increase spawning substrate availability (Pander *et al.*, 2015). Furthermore, in-stream gravel cleansing techniques, such as substrate raking, trigger mechanical bioturbation of sediments, promoting fine sediment mobilisation and improved alterations of spawning gravels for lithophilic fishes (Pander *et al.*, 2015).

1.5.5 Macroinvertebrate Community Structure

Macroinvertebrates support a range of ecosystem functions, including the breakdown of organic matter and consumption of algae and biofilms to support energy transfers through the food chain. In addition, they facilitate the downstream movement of organically bound toxicants (Wallace and Webster, 1996). Macroinvertebrate functional groups – grazers, shredders, gatherers, filterers and predators utilise morpho-behavioural mechanisms in order to obtain food sources (Wallace and Webster, 1996). In addition, invertebrates influence important processes and biological functions including nutrient cycling, primary production and decomposition. Invertebrate groups include: scrapers - which scrape material including algae and periphyton from minerals and organic matter; shredders - which consume coarse decomposing plant tissue, wood or microflora/fauna; gatherers - which feed on FPOM; filterers which remove suspended particle matter from water and provide more palatable material for consumers through their faeces; and

predators - which feed on animal matter. Functional macroinvertebrate groups are adapted to perform specialised ecosystem functions. For example, stoneflies shred and consume material (selecting leaves microbially conditioned by colonising fungi and bacteria), while flattened mayflies and snails help to control energy up the food chain through the consumption of periphytic algae and biofilms (Wallace and Webster, 1996; Wotton and Malmqvist, 2001).

Macroinvertebrates are useful indicators of stream health. Their sedentary nature allows for prolonged exposure to chemical conditions present in sediment and water (Miller and Kochel, 2010). Furthermore, their vulnerability to environmental changes allows for the interpretation of environmental conditions (including available benthic resources) and pollution events (Everaert *et al.*, 2014). Physical factors including relief and lithology place several constraints on invertebrate habitats and food availability (Wallace and Webster, 1996). Invertebrates are a critical food resource for littoral, pelagic fish and birds and therefore exposure to poor sediment and water quality can influence species development and functional feeding patterns (Woodward and Warren, 2007). In urban streams, invertebrates are commonly affected by inputs of inorganic nitrogen and metals which can reduce biotic diversity and increase the dominance of tolerant species (Grimm *et al.*, 2005; Wollheim *et al.*, 2005). The lifespan of invertebrates, although shorter than that of fish, can reflect environmental conditions and highlight a graded response to acting pressures (Visser *et al.*, 2017). Some studies have found that elevated agricultural runoff can reduce leaf litter breakdown rates through reductions in *G. pulex* from N pollution (Piscart *et al.*, 2009). Other studies examining the response of Asellidae spp. and *Gammarus pulex* to toxicants being expelled from stormwater overflows found that aqueous copper, dissolved copper and zinc were responsible for reductions in individuals present (Mulliss *et al.*, 1996). Pesticides can influence feeding and survival of *G. pulex*

and other macroinvertebrates, resulting in lowered degradation rates of leaf litter (Nyman *et al.*, 2013). Nonetheless, some species can also successfully reproduce and recolonise under these conditions. Studies have found that restoring shredder functional groups following previous damage from insecticides resulted in restored leaf litter breakdown processes (contributing to 25-28% of annual processing) and export of FPOM (Cuffney *et al.*, 1990).

1.6 Chapter Summary

This chapter explored the negative effects of human activity on ecosystem structure and functions in urban streams. Subsequently, it investigated the importance of nitrogen processing and decomposition as controls on ecosystem structure (fish and macroinvertebrate assemblages). Few studies have investigated the mechanisms by which different restoration activities may elicit positive ecosystem responses. This thesis seeks to explore these structures and functions through assessments of nitrogen dynamics, decomposition and ecological community structure.

Chapter 2: Study Sites

Five paired restored and unrestored sites from urban tributaries of the River Thames in Greater London were selected for this project from the River Restoration Centre (RRC) database and from previous works by Smith and Chadwick (2014). The rivers selected were the Ravensbourne, Pool, Hogsmill, Wandle and Brent (Figure 2.1; Table 2.1). The sites comprised 25 metre (m) long reaches which varied in terms of urban cover, land use and restoration practice (Smith and Chadwick, 2014). On the river Brent and Wandle, the restored reach was downstream, whereas on the Pool, Ravensbourne and Hogsmill the restored reach was upstream. Paired reaches were approximately 50 – 250 m apart across all reaches.

Geomorphological features along these reaches included low gradient and shallow beds (<0.5 m), non-turbulent flows and underlying geology dominated by chalk. 85% of the world's chalk streams are located in England, of which many are concentrated in the south-east (Salter and Singleton-White, 2019). Chalk streams are fed by mineral-rich streams which are characterised by clean waters, stable temperatures and regulated flows. The alkalinity of chalk streams provides a critical habitat for macroinvertebrate and fish species (Salter and Singleton-White, 2019).

Land use across these sites is predominantly urban, owing to high density housing within each catchment boundary. Historic channel straightening, culverting and industrial activities (e.g. mills) have historically led to concerns over flooding, contamination and hydrologic connectivity along these river networks (Smith and Chadwick, 2014; Cook, 2015).

Collectively, restoration efforts within the study rivers (Ravensbourne, Pool, Wandle, Hogsmill and Brent) have primarily focused on restoring heterogeneous flows, hydrological connectivity and habitat biodiversity (Table 2.1).

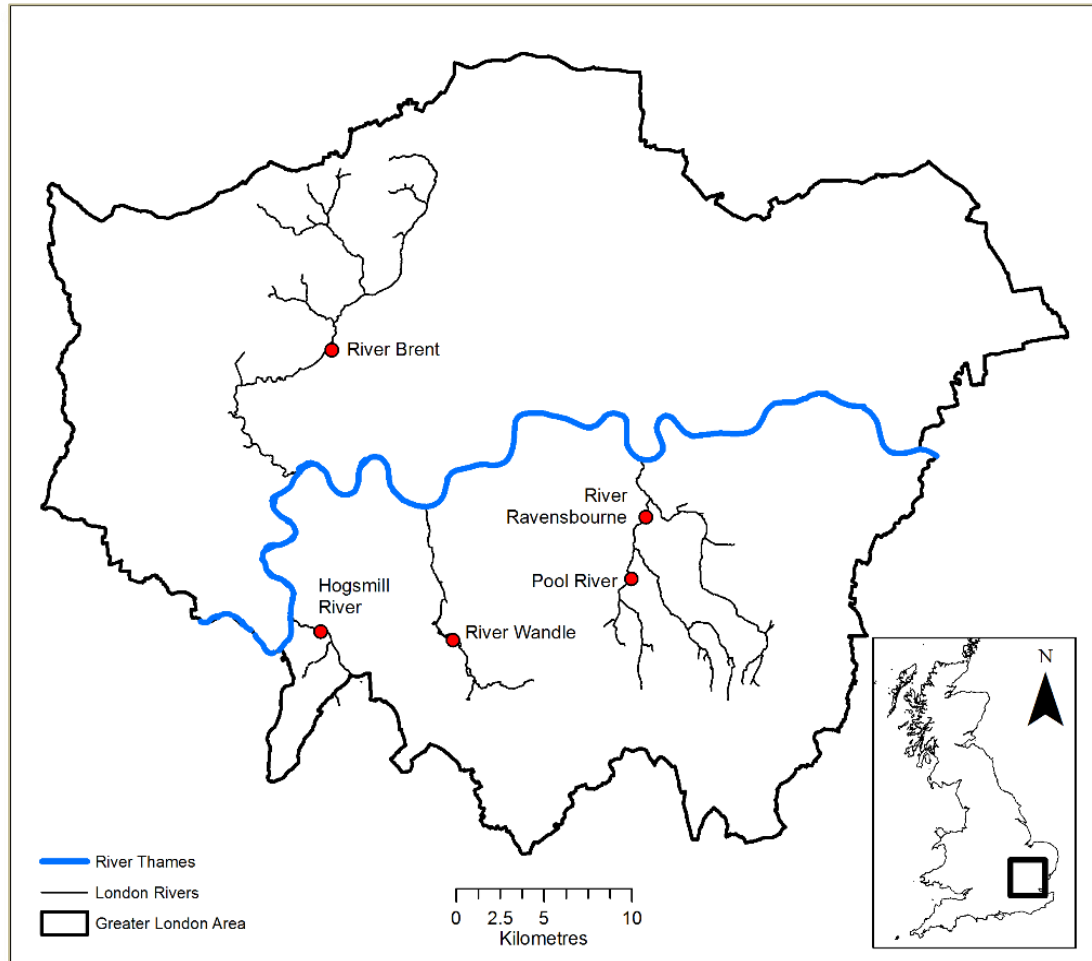


Figure 2.1. Study sites situated within Greater London, UK. Dots highlight the locations of each of the five study rivers (Ravensbourne, Pool, Wandle, Hogsmill and Brent).

Table 2.1. Characteristics of restoration among the study rivers, including the upstream location of restored and unrestored reaches, year of restoration completion, river length (km) and % urban cover at the study rivers. Sites comprised 25 m long reaches which were assessed in this project.

River	Site	Restored U/S Location	Unrestored U/S Location	Year Completed	Length	% Urban	Urban pressures	Restoration
Ravensbourne	Ladywell Fields	TQ3741974352	TQ3767574519	2008	17.4	51	Channelisation & culverting	Remeandering through parks
Pool	Bell Green	TQ3721572840	TQ3726472993	2012	5.1	57	Culverting, vegetation & fish loss	Berms & redirecting flows
Wandle	Carshalton	TQ2815565015	TQ2802364814	2015	14	47	Impoundment, weirs, low flow & oxygen levels	Lowering of weir & shortening fish passages
Hogsmill	Green Lane	TQ2141063354	TQ2115163524	2014	9.9	39	Fish pass obstructions, weirs & sewage	Weir removals, creation of pools & riffles, channel narrowing
Brent	Tokyngton Park	TQ1998384583	TQ2010184715	2003	29	69	Impoundments & habitat degradation	Recycling of concrete, remeandering & creation of backwaters

2.1 River Ravensbourne: Ladywell Fields

The River Ravensbourne is 17.4 km in length and has a total catchment area of 180 km² (Lewisham Council, 2010). The river originates in Caesar's Well (Kreston) and runs through the Boroughs of Bromley, Lewisham and Greenwich to meet the River Thames at Deptford Creek. The River Ravensbourne at Ladywell Fields, which formerly ran along the edge of the park, had been artificially widened and toe-boarded at its banks. A large proportion of the river was hidden behind railings and dense vegetation, reducing its societal and ecological value (Figure 2.2).

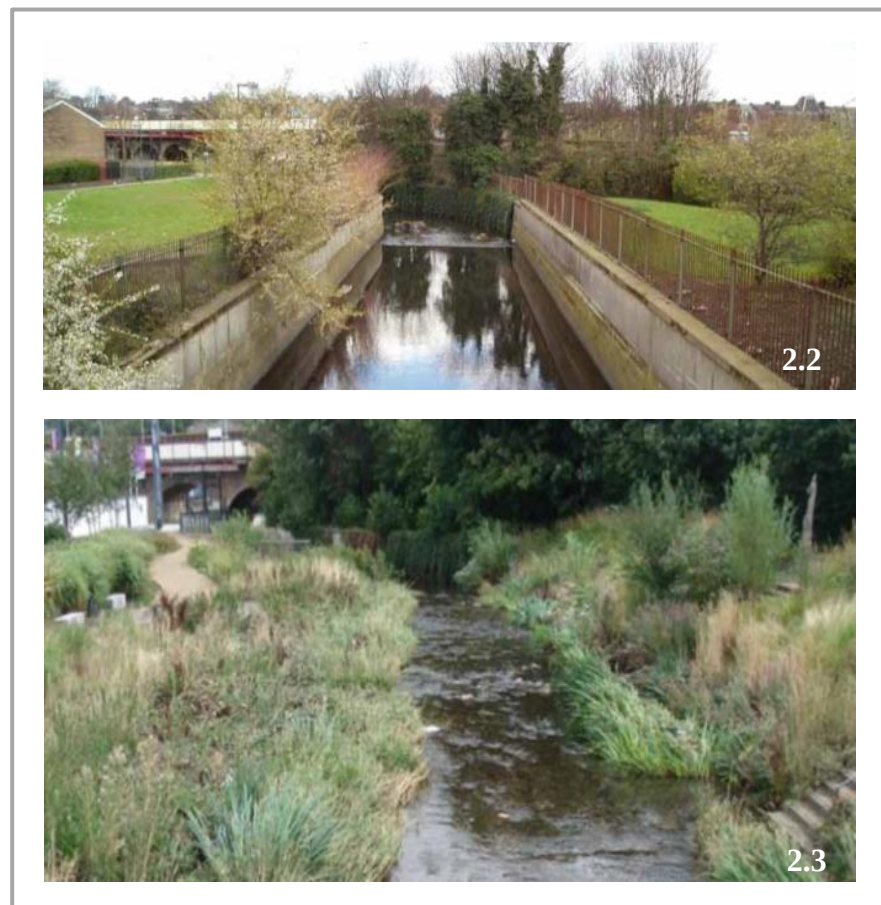


Figure 2.2. Photograph of a degraded reach of the River Ravensbourne prior to restoration; **Figure 2.3.** Photograph of a restored reach of the River Ravensbourne following culvert liberation and remeandering (both obtained from Lewisham Council, 2010).

Restoration works for the River Ravensbourne at Ladywell Fields (finish date 2008, length 200 m, cost £400,000) were proposed as part of QUERCUS (Quality Urban Environments for River Corridor Users and Stakeholders), a partnership between the London Borough of Lewisham and Chester City Council (Lewisham Council, 2010). Partner organisations included the Environment Agency, Building Design Partnership, Arup and Fergal Contracting. The key aims of the restoration works were to liberate the culverted river, divert flow and enhance structural diversity by remeandering the channel through the centre of the park (Figure 2.3). The park was intended to provide an area for flood water storage, further enhanced by the creation of backwaters downstream of the park footbridge and an ephemeral pond to provide refugia for fish and bird species. The River Restoration Centre (2008a) highlighted the positive outcomes associated with the project, including improvements to recreational value and species (fish and bird) diversity. The restored reach examined in this study was located just upstream of the park. In-stream features included woody debris, designed to mediate flows (figure 2.4).

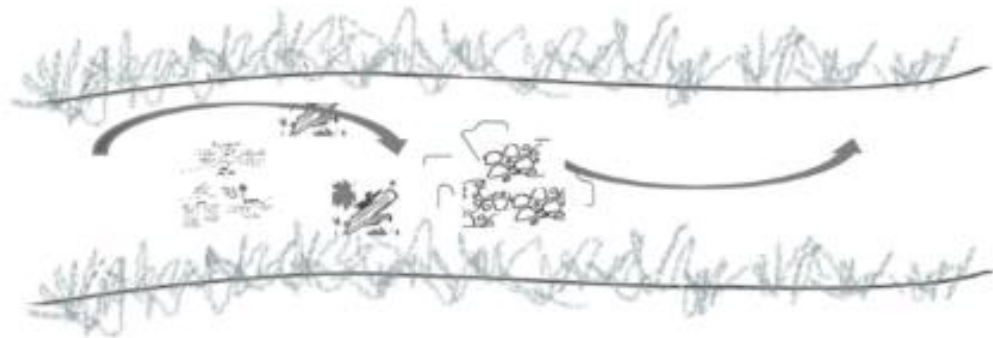


Figure 2.4. Schematic of the restored River Ravensbourne site from upstream (left) to downstream (right). The site was characterised by marginal vegetation, shallow beds, in-stream woody debris and entrapment of gravels and fine sediment. There were observed riffles owing to the shallow bed habitat and in-stream gravels.

2.2 River Pool: Bell Green

The River Pool, a tributary of the River Ravensbourne, is 5.1 km (3 miles) in length. It rises between Shirley and West Wickham in the London Borough of Croydon. The river flows northwards through Beckenham in the borough of Bromley and through Sydenham in the London Borough of Lewisham to the Ravensbourne (Catford). A large area of the Pool lies within a floodplain. Historically the reach near Sydenham which passed through Bell Green gasworks was canalised to mitigate against the possible effects of gas intrusion. Large proportions of the Pool were historically straightened (Figure 2.5).



Figure 2.5. Photograph of the River Pool at the unrestored site examined in this project (obtained from Talling, *London's Lost Rivers*); **Figure 2.6.** Photograph of the restored River Pool site located ~200 m upstream, showing installations of spoil to encourage flow sinuosity (obtained from Lewisham Council, 2010).

Prior to restoration works, this stretch of river was largely featureless with restricted vegetation present. In 1994, berms were deployed in-stream and flows redirected to elicit flow sinuosity and encourage habitat heterogeneity (RRC, 2012). More recent work was undertaken by Thames 21 between 2011 and 2012 along a 300 m stretch of river. Site-won wood and spoil from the Environment Agency were deployed in-channel to encourage flow sinuosity (Figure 2.6; 2.7). Additional consideration was made to manage invasive species including Japanese Knotweed, Himalayan Balsam and Giant Hogweed when narrowing down the channel.

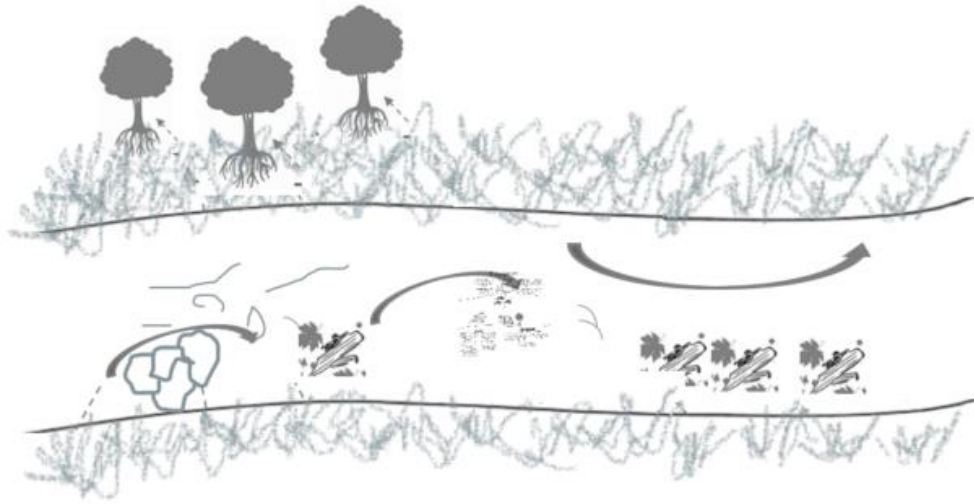


Figure 2.7. Schematic of the restored River Pool site from upstream (left) to downstream (right). The site was characterised by marginal vegetation, in-stream boulders, woody debris. Flow was characterised by riffles and glide.

2.3 River Wandle: Carshalton

The River Wandle stretches 14 km from the lower dip slope of the North Downs to the main River Thames via Croydon, Sutton and Merton. In the eighteenth century, the Wandle was a flourishing trout stream, but increasingly became subject to sewage pollution, over-abstraction and channelisation (Cook, 2015; South East Rivers Trust, 2010,'11,'15,'15a,'15b) (Figure 2.8). The implementation of inadequate fish passages and barriers have impeded upon longitudinal connectivity (Pike *et al.*, 2014). Combined with storm water inputs from sewage works, this has triggered sediment deposition, nutrient loading and oxygen depletion (Pike *et al.*, 2014).



Figure 2.8. Photograph of the unrestored River Wandle site at Carshalton (unrestored reach (obtained from the South East Rivers Trust); **Figure 2.9.** Photograph of restored River Wandle site near Butter Bridge (author's own).

The Carshalton branch of the River Wandle has undergone several phased projects to restore channel connectivity and habitat diversity. Phase 1 took place between 2010 and 2011 with the lowering and removal of three weirs. In addition, a Larinier pass was added to a fourth weir to improve fish passage brown trout (*Salmo trutta*) (South East Rivers Trust, 2010). Phase 2, (January-October 2011), involved notching a small weir at Millpond place and replacing the impounded section with a narrower channel to provide resilience to variable flows (South East Rivers Trust, 2011). Phase 3 was undertaken between June 2012 and March 2015. This involved reducing the height of the weir at Butter Hill by 1 m to increase efficient transport. A further 500 m of river (assessed in this study) was remeandered with features including berms, riffles and marginal wetlands to encourage improvements in habitat diversity (Figure 2.9; 2.10). Additional work was undertaken during this time to manage urban diffuse pollution. Rain gardens, downstream defenders, mycofilters (substrate), siltex (chalk-like natural material) and smart sponges (a product to absorb oil) were installed to enhance the removal of pollution and promote natural stream conditions. (South East Rivers Trust, 2015). Annual stocking of common dace *Leuciscus leuciscus*, European chub *Leuciscus cephalus*, roach *Rutilus rutilus* and common barbel *Barbus barbus* have improved ecological community structure along this river.



Figure 2.10. Schematic of the restored River Wandle site from upstream (left) to downstream (right). The left bank (looking downstream) was enclosed by an artificial wall which protected the adjacent road. The site was characteristically narrow, shallow and meandering, fostering high energy flows. There was a high density of in-stream vegetation and macrophytes, providing a refuge habitat for ecological communities. Glide was the main flow type observed at this site.

2.4 Hogsmill River: Green Lanes

The Hogsmill River is 9.9 km in length, it rises in Ewell and flows through the district/borough of Epsom, Ewell and Kingston upon Thames (South East Rivers Trust, 2013, '14,'18). The river has five tributaries: Green Lanes Stream, Ewell Court Stream, Horton Stream, Tolworth Brook and Hogsmill Stream. The Hogsmill River has previously been channelised and subject to significant pollution from stormwater tanks (Figure 2.11).



Figure 2.11. Photograph of the restored Hogsmill River site at Green Lanes; **Figure 2.12** Photograph of the unrestored Hogsmill River site at Green Lanes.

The Hogsmill River at Green Lanes Stream has undergone several phased projects to restore connectivity and enhance habitat diversity. The Hogsmill River Connectivity Project which took place between 2012 and 2013 involved removing high head weirs (400 m apart) to restore fish passage and enhance biodiversity. In total, 140 m of

revetment walls were removed. Additional work was conducted between 2012 and 2014 to renaturalise the surrounding banks for wildlife (Figure 2.12; 2.13). 200 tonnes of concrete were removed from the banks and engineered in the river to encourage sinuosity of flows (South East Rivers Trust, 2013). Between 2017 and 2018, the Hogsmill Bridge Footings project removed three weirs to allow for multiple fish species to pass through under low flow conditions (South East Rivers Trust, 2018). Following this, the River Club Weir Removal (July 2016-May 2017), supported by the Environment Agency, Thames Rivers Trust and private landowners, included removing the River Club Weir, and removing ~100 m of impoundments upstream. Approximately 35 tonnes of concrete and steel were removed (from inside the channel) and replaced with chestnut posts and hazel faggots. Biodegradable and coir geotextile material was covered over this (South East Rivers Trust, 2017).

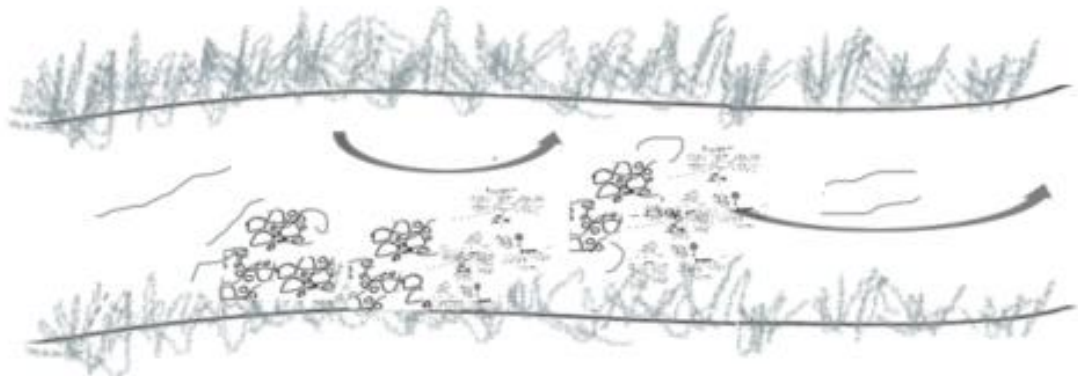


Figure 2.13. Schematic of the restored Hogsmill River site from upstream (left) to downstream (right). The site was characteristically narrow and shallow. There were observed riffles owing to the shallow bed habitat and in-stream gravels.

2.5 River Brent: Tokyngton Park

The River Brent, situated in west and northwest London is 29 km in length. The river rises in the Borough of Barnet, flowing through Wembley until it reaches the main River Thames at Brentford. Extensive historical flood alleviation works undertaken in the 1940's and 1970s led to this section of river being straightened and encased in concrete (Figure 2.14). River Restoration at Tokyngton Park, Wembley finished in 2003 and spanned a length of 2 km (River Restoration Centre, 2008b). Project partners included the London Borough of Brent and the Environment Agency. The project was designed with the aim of removing concrete banks and creating an attractive space for the public and wildlife. Concrete was removed from the banks and stabilised using willow poles, crushed concrete and silt (Figure 2.15; 2.16) Manipulated bed levels were created with the aim of fostering pool and riffle sequences. Additionally, backwaters were created with reeds to provide shelter for fish during flood episodes (River Restoration Centre, 2008b). The Brent is prone to pulses of pollution from misconnected pipes (Smith and Chadwick, 2014).

At this study reach, flood and pollution preventative approaches have been taken to deploy willow poles, re-cycle ground concrete to generate riffle-pools and encourage habitat stabilisation. The creation of backwaters has led to the succession of new habitats, acting as a buffer zone during pollution and flood events (River Restoration Centre, 2008b).



Figure 2.14. Photograph of the River Brent unrestored site at Tokyngton Park; **Figure 2.15.** Photograph of the River Brent restored site at Tokyngton Park.

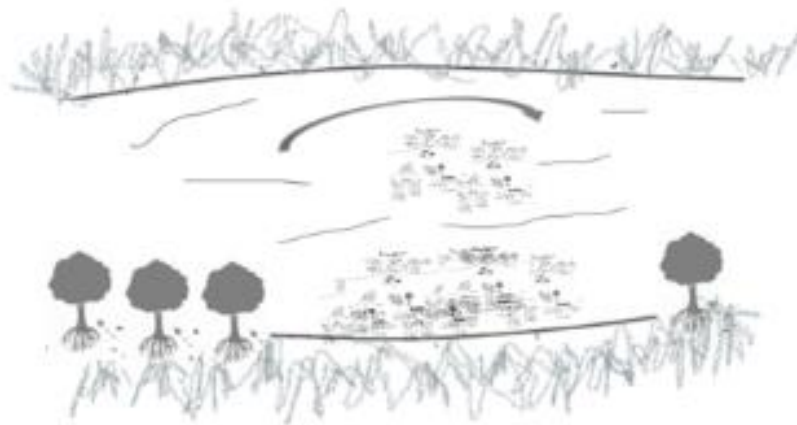


Figure 2.16. Schematic of the restored River Brent site from upstream (left) to downstream (right). The channel was characteristically wide, deep and dominated by fine sediments. There was significant evidence of bank scour, evident of unrestricted high-energy flows. In areas of low energy, a significant amount of fine sediment had been deposited. The flow type varied between glide and run.

Chapter 3: Influence of Urban River Restoration on Nitrogen Dynamics at the Sediment-Water Interface

Abstract

River restoration projects focused on increasing flow heterogeneity and channel biodiversity through use of in-channel structures can facilitate ecosystem services, such as promoting nitrogen (N) storage to reduce eutrophication. In this study, flux chambers were used to examine ammonium (NH_4^+) and nitrate (NO_3^-) cycling across the sediment-water interface. Paired restored and unrestored study sites across 5 urban tributaries of the River Thames, Greater London were used to examine N dynamics following physical disturbances (0-3 min exposures) and subsequent biogeochemical activity (3-10 min exposures). Average ambient NH_4^+ concentrations were significantly different amongst all sites and ranged from 28.0 to 731.7 $\mu\text{g l}^{-1}$, with the highest concentrations measured at restored sites. Average NO_3^- concentrations ranged from 9.6 to 26.4 mg l^{-1} but did not significantly differ between restored and unrestored sites. Average NH_4^+ fluxes at restored sites ranged from -8.9 to 5.0 $\mu\text{g N m}^{-2} \text{ sec}^{-1}$, however restoration did not significantly influence NH_4^+ uptake or regeneration (i.e. a measure of release to surface water) between 0-3 minutes and 3-10 minutes. Further, average NO_3^- fluxes amongst sites responded significantly between 0-3 minutes ranging from -33.6 to 97.7 $\mu\text{g N m}^{-2} \text{ sec}^{-1}$. Neither NH_4^+ nor NO_3^- fluxes correlated to sediment chlorophyll-*a*, total organic matter or grain size. Variations in overall N fluxes were attributed to N-specific sediment storage capacity, biogeochemical transformations, potential legacy effects associated with urban pollution and variations in river-specific restoration actions. A greater understanding of mechanisms driving the dynamic transfer, processing and removal of NH_4^+ and NO_3^- in urban river systems is required for informing future river restoration strategies.

3.1 Introduction

The “urban stream syndrome” provides a framework for evaluating changes associated with urbanisation (Paul and Meyer, 2001; Meyer *et al.*, 2005; Walsh *et al.*, 2005; Booth *et al.*, 2016; Vietz *et al.*, 2016), including physical habitat modifications, hydrological alterations and elevated nutrient loads occurring in catchments across the globe (Vörösmarty *et al.*, 2010; Vilmin *et al.*, 2018). In urban environments, impervious surface cover and channel impoundments can off-set hydrologic connectivity between the stream channel, hyporheic, and riparian zones, resulting in complex sediment-supply dynamics (Chadwick *et al.*, 2006; Harrison *et al.*, 2011). In addition, altered flow regimes can modify ecological function (Vörösmarty *et al.*, 2010), including nutrient cycling (Kaye *et al.*, 2006; Kaushal and Belt, 2012; Reisinger *et al.*, 2016), which can be compounded by elevated nutrient loads from gutters and storm drains (Rueda *et al.*, 2002; Bernot and Dodds, 2005; Meyer *et al.*, 2005). ‘Urban karsts’, encompassing a complex, predominantly hidden, network of buried headwater streams, sewers, and potable water pipes can further modify hydrological processes, reducing water infiltration and inhibiting nutrient storage capacity (Kaushal and Belt, 2012; Pennino *et al.*, 2014). Together these factors can play a major role in influencing nitrogen dynamics in urbanised river ecosystems.

The presence of nitrogen in urban rivers is a major management issue due to high inputs from runoff and groundwater contamination (Grimm *et al.*, 2005). Recent studies have estimated that anthropogenic N from grey water footprints can contribute up to 32.6 million tonnes per year to freshwater systems (Mekonnen and Hoekstra, 2015), resulting in widespread problems with eutrophication and hypoxia (Mulholland *et al.*, 2008). In addition, urban watersheds receive N inputs from indirect sources, such as atmospheric deposition, diffuse land-based practices (e.g. fertilisers), unregulated discharges, leaky septic pipes and misconnections (Craig *et al.*, 2008; Kaushal *et al.*, 2011; Kaushal and

Belt, 2012). N enters streams via runoff, interflow, groundwater or artificial drainage and the rate and mode of this is controlled by several factors including geology, soil type and land use. Nitrate is the most dominant form of N, owing to its high solubility and leaching capacity. In urban streams in areas such as London, runoff transport is high owing to high impervious cover, a trigger for elevated concentrations of N. Wollheim *et al* (2005) stated that there is low biological N removal through runoff delivery. Ammonium is less prevalent as it is immobilised easily, adsorbs to clay particles and organic matter (negatively charged) and is often nitrified in small streams (Craig *et al.*, 2008). In the Thames catchment, NO_3^- concentrations have been reported in ranges between ~5 to ~35 mg l^{-1} (Neal *et al.*, 2006a; Davies, 2011) while NH_4^+ has been noted between ~100 to ~700 $\mu\text{g l}^{-1}$ (EEA, 2017; Pecorelli, 2017). These concentrations from highly urban environments differ significantly from lower N concentrations observed in more rural UK rivers ($<100 \mu\text{g l}^{-1}$) (EEA, 2017).

Nitrogen dynamics (including uptake, regeneration and denitrification) which occur via physical and biogeochemical processes, are influenced by a wide range of factors including river discharge, sediment type, water quality and stream metabolism (Lijklema *et al.*, 1993., Valett *et al.*, 1996; Clavero *et al.*, 2000; Maksymowska-Brossard, 2001; Wu and Yang, 2012). NO_3^- is highly abundant in urban rivers and subject to assimilation, storage and denitrification via algae, aquatic plants and microbes (Grimm *et al.*, 2005; Kaushal *et al.*, 2008; Pennino *et al.*, 2014). N is also known to control and limit Chl-*a* concentrations in urban systems (Neal *et al.*, 2006b) and is influenced by sediment type and quality and quantity of organic matter (Kaushal *et al.*, 2011). N retention can occur via temporary storage or permanent removal (Newcomer Johnson *et al.*, 2016; Figure 3.1). Temporary storage refers to biological retention (e.g. microbes, algae and vegetation) which assimilate and store N until it is released by remineralisation or

decomposition (Craig *et al.*, 2008). Physical removal occurs in areas of reduced water flow where ground and surface waters mix (e.g. hyporheic zones) (Ensign and Doyle, 2005). N removal can occur along groundwater flowpaths through denitrification, transport modes, notably groundwater transport, however impairments of regular hydrological conditions (e.g. culverted streams) may limit removal capacity. Plants, fungi and bacteria can temporarily assimilate N which is further increased through sunlight exposure and increased retention times (Newcomer Johnson *et al.*, 2016). By comparison, permanent removal occurs through coupled nitrification-denitrification (Craig *et al.*, 2008; Newcomer Johnson *et al.*, 2016; Figure 3.1). This is a two-step process whereby compounds including NH_4^+ , NO_2^- and NO_3^- are converted into inert gaseous products (N_2O and N_2) under certain conditions. Ammonium is firstly oxidised to nitrite and then converted to nitrate through nitrification ($\text{NH}_4^+ \rightarrow \text{NO}_2^- \rightarrow \text{NO}_3^-$) which occurs under aerobic conditions. Secondly, denitrification, which is microbially mediated and necessitates anoxic conditions and an electron donor (e.g. organic carbon), reduces nitrate to nitrous oxide and nitrogen gas ($\text{NO}_3^- \rightarrow \text{N}_2\text{O} \rightarrow \text{N}_2$) (Newcomer Johnson *et al.*, 2016). Coupled nitrification and denitrification can be enhanced by oxygenated root surfaces in saturated soils, while macrophytes can also retain N (Newcomer Johnson *et al.*, 2016). Denitrification occurs more readily in areas with features such as debris dams or organic rich sediments. The physical and biogeochemical processes that influence such N dynamics are predominantly focused at the sediment-water interface and more investigations surrounding these ecosystem functions are needed in urban rivers (Grimm *et al.*, 2005).

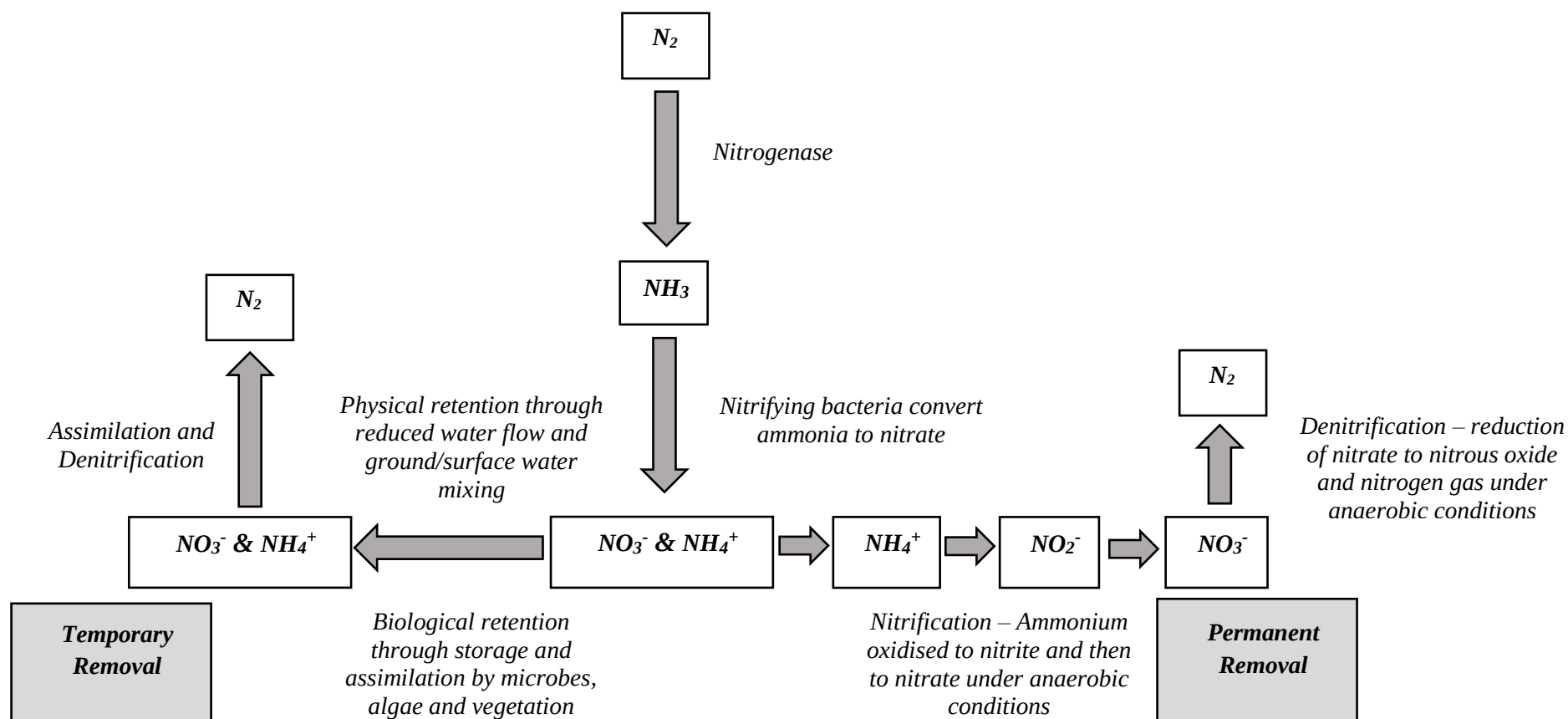


Figure 3.1. Summary of aquatic ecosystem biogeochemical cycling, illustrating processes of N transfer in aquatic systems. Schematic showing nitrate (NO_3^-), ammonium (NH_4^+), nitrite (NO_2^-) and N_2 (nitrogen) processing.

Secondly, denitrification, which is microbially mediated and requires anoxic conditions and an electron donor such as organic carbon, reduces nitrate to nitrous oxide and nitrogen gas ($\text{NO}_3^- \rightarrow \text{N}_2\text{O} \rightarrow \text{N}_2$).

To date, a handful of studies have examined the implications of river restoration on N processing (Bernhardt *et al.*, 2005; Bernhardt *et al.*, 2007; Kaushal *et al.*, 2008). Traditional assessments of stream health characterise structural condition (e.g. fish and invertebrate community), rather than functional process (e.g. primary production, nutrient cycling and decomposition) (Tiegs *et al.*, 2007; Young *et al.*, 2008; Tiegs *et al.*, 2013). Most restoration practices have focused on improving hydromorphology rather than modifying biogeochemical processes (Bernhardt *et al.*, 2007). However, recent approaches have considered how habitat engineering focused on geomorphic stabilisation, hydrologic connectivity and flow manipulations (e.g. creating debris dams, backwaters and eddies) can influence N dynamics (including nitrification and ammonification) via uptake and regeneration (Roberts *et al.*, 2007; Craig *et al.*, 2008; Wohl *et al.*, 2015). Additionally, modifying flow regime can encourage sediment organic matter retention and hyporheic anoxia due to increased heterotrophic respiration and prolonged contact time with denitrifying bacteria (Kasahara and Hill, 2006; Bukaveckas, 2007; Craig *et al.*, 2008). Further links have also been made between restoration activity, uptake lengths (Hines and Hershey, 2011) and increased N ion retention capacities, which can result in nutrient reductions further downstream (Webster *et al.*, 2003; Grimm *et al.*, 2005).

Craig *et al.* (2008) reviewed stream strategies for reducing N loads through several approaches including: increasing carbon availability; increasing contact with benthos; and increasing connectivity between streams and adjacent environments. Authors have reported on rapid removal of N at sites where biological demand is high (Hall and Tank,

2003; Webster *et al.*, 2003). Given that carbon availability may be a limiting factor for assimilation and denitrification (Groffman *et al.*, 2005), implementing structures which allow for these mechanisms to be carried out may support the removal of N. In addition, promoting contact with benthos allows for increased topographic complexity and surface area to volume ratios, resulting in increased water-benthos interactions. Features such as woody debris and rifle pools can encourage this removal. Newcomer Johnson *et al.* (2016) conducted a review analysing nutrient retention within hydrologically reconnected streams. Results showed that nitrate uptake metrics were significantly related to watershed surface area, impervious surface cover and average reach width ($p < 0.05$). Ammonium uptake metrics were significantly related to discharge, velocity and transient storage ($p < 0.05$), owing to possible restoration practices including floodplain connectivity triggering N retention.

Due to the need for greater understanding of N biogeochemical processes following river restoration, the *aim* of this study was:

‘To examine whether restoration influences nitrogen processing across the sediment-water interface in urban tributaries of the River Thames, London’.

The objectives were:

- 1:** To examine ambient nitrogen (nitrate and ammonium) concentrations and further determine whether nitrogen uptake (net movement into sediment) and regeneration (i.e. a measure of release to surface water) varies between restored and unrestored sites.
- 2:** To quantify the effect of physical and biogeochemical activity on nitrogen uptake and regeneration.
- 3:** To investigate relationships between chlorophyll-*a*, organic matter, sediment size distribution and nitrogen processing.

Furthermore, it was hypothesised that:

1: Restoration would reduce nitrate and ammonium concentrations and facilitate increased nitrogen uptake at restored sites compared to unrestored sites, owing to the presence of in-stream features driving nitrogen assimilation.

2: Physical disturbances and biogeochemical activity would increase regeneration at unrestored sites due to sediment nutrient saturation and increase uptake at restored sites due to higher sediment sink capacities.

3: Higher concentrations of chlorophyll-*a* and proportions of organic matter would be linked to nitrogen removal due to facilitated assimilative processes, while higher proportions of fine sediment would be linked to nitrogen regeneration due to an increased sediment storage capacity.

Overall, it was hypothesised that restoration would improve nutrient storage capacity through biological and physical retention. This was achieved through the examination of sediment-water interface assays to quantify NH_4^+ and NO_3^- fluxes as uptake from the water column into the sediment, or regeneration from the sediment into the water column in restored and unrestored tributaries of the River Thames, Greater London, UK.

3.2 Materials and Methods

3.2.1 NH_4^+ and NO_3^- Flux Assays

At each reach during four sampling events in Spring 2016 (March-May), 20 random patches were selected and 10 ml of fine surficial sediment (top 2 cm of stream bottom) was collected with a stainless-steel scoop. The ‘patch-scale’ is generally considered to reflect substratum characteristics while the ‘reach-scale’ is considered to reflect channel topography (Lepori *et al.*, 2005). Patch-scale studies are important because they represent a mosaic of processes occurring at the reach-scale. Ambient water samples (grab samples taken from the downstream end of each reach) were also obtained at all sites, filtered (0.22 μm mixed cellulose ester membrane filters), transported back to the laboratory and

stored at -20°C . NH_4^+ and NO_3^- analysis was conducted subsequently, using the method described below.

Sediments collected from each random patch were transferred into 50 ml tubes and mixed with 35 ml stream water (Figure 3.2). For NH_4^+ analysis, 2.5 ml water was extracted ($T = 0$) and again after 3 ($T = 3$) and 10 minutes ($T = 10$). The initial 0-3-minute flux was equated to physical disturbance events (e.g. sediments disturbed by a rising flood flows) and expected release of N from the sediment sample into the overlying water column. The 3-10-minute flux was then equated to biogeochemical flux which could mimic the movement of N between the water column and sediment layers due to biogeochemical processes. Based on a pilot study, these two sampling periods reflect the time required for sediment particles to settle ($T = 0\text{-}3$ min) and where water temperature would not be affected by air temperature (e.g. reflecting air temperature affects biogeochemical processes; $T = 3\text{-}10$ min) (Figure 3.2) The 2.5 ml water sample for NH_4^+ analysis was added to 10 ml working reagent (containing 2 l borate buffer, 10 ml sodium sulphite and 100 ml ortho-phthalaldehyde solution) in a separate vial and analysed using fluorometric methods (Holmes *et al.*, 1999). An additional 7.5 ml water sample was filtered ($0.22\text{ }\mu\text{m}$ mixed cellulose ester membrane filters), transported back to the laboratory and stored at -20°C . Subsequently, samples were thawed, filtered ($0.22\text{ }\mu\text{m}$ mixed cellulose ester membrane filters) and NO_3^- concentrations determined using ion chromatography. Due to the field-based nature of these assays, a few samples were not suitable for analysis, resulting in 12–20 replicates per reach with a final sample size of 158 successful assays completed.

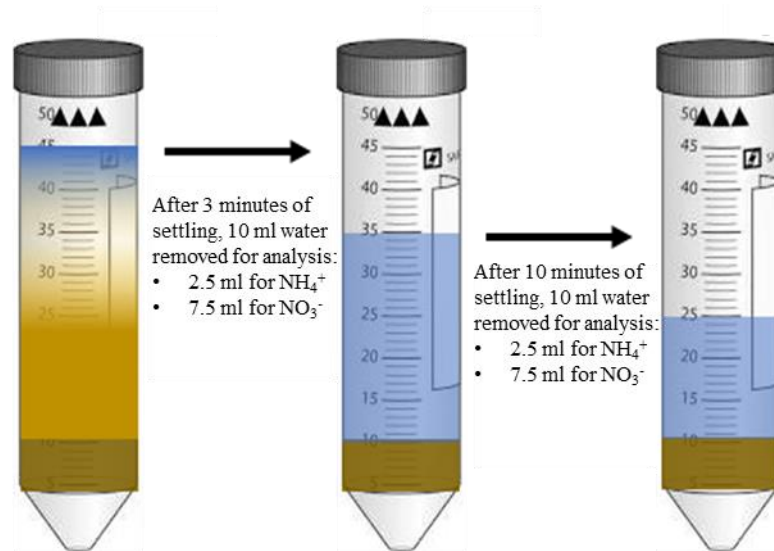


Figure 3.2. Experimental flux chambers: 10 ml sediment from the benthic zone was randomly collected, transferred into separate 50 ml falcon tubes and mixed with 35 ml stream water. For N samples, 10 ml water (2.5 ml for NH_4^+ and 7.5 ml for NO_3^- analysis) was extracted after the sediment had settled ($T = 0$ minutes) and after both 3 ($T = 3$ minutes) and 10 minutes ($T = 10$ minutes). The initial 0-3-minute flux represented a “physical” disturbance event, while the 3-10-minute flux reflected a “biogeochemical” flux.

3.2.2 Sediment Analysis

Sediment grain size analysis was carried across all sites. Distributions were determined from 5 separate 10 g fine benthic sediment subsamples collected from both the restored and unrestored reaches of the study streams. Samples were dried (>24 hours at 60°C), weighed and sieved to separate coarse (>1 mm) and fine sediment (<1 mm). Sediment was dispersed into a Malvern Mastersizer 2000 granulometer and examined for average particle size. This procedure was repeated three times for each subsample. Samples were classified as either sand (0.063-2 mm), silt (0.004-0.063 mm) or clay (<0.004 mm).

After measuring N fluxes, sediment samples were mixed with 10 ml methanol for 1 minute and left in the dark for an hour to extract Chl-*a*. A 1.5 ml of the supernatant was transferred into an Eppendorf tube and centrifuged for one minute at 3000 rpm. The absorbance of the sample was measured at 665 and 750 nm Abs to account for Chl-*a*

extracted and background turbidity (Marker, 1994). Chl-*a* concentrations were calculated and expressed as μg^{-1} Chl-*a* g^{-1} dry weight using the following equation:

$$\frac{13.9 [Abs_{665} - Abs_{750}] * vol\ extracted\ (ml)}{Sediment\ mass(g)}$$

Equation 3.1.

Where Abs_{665} and Abs_{750} refer to absorption at respective wavelengths; volume extracted (ml) and sediment mass (g).

For % total organic matter (TOM) sediment samples used for the Chl-*a* measurement were dried in an oven at 60°C for 24 hours. Samples were subsequently transferred into crucibles and weighed prior to and after ashing at 550°C for 6 hours. TOM was measured as a percentage of weight loss on ignition and did not include the TOM associated with the extracted Chl-*a*.

3.2.3 Data Analysis

N fluxes were derived from the following equation:

$$\frac{[N_2] - [N_1]}{A * [t_2 - t_1]}$$

Equation 3.2.

Where N_2 and N_1 refers to the NH_4^+ or NO_3^- concentrations at t_2 and t_1 , respectively; A is the surface area of the sediment surface (m^2) and $t_2 - t_1$ = the time (sec) between the subsequent (t_2) and previous (t_1) water sample. NH_4^+ and NO_3^- fluxes are expressed as $\mu\text{g N} / (\text{m}^2 * \text{sec})$. A positive flux indicates the movement of N from the sediment into overlying waters and a negative flux indicates the movement of N from overlying waters into the sediment.

Average NH_4^+ and NO_3^- concentrations and fluxes, Chl-*a* concentrations and % TOM were compared between restored and unrestored sites on each river and between rivers using a two-way ANOVA on ranks, followed by a Tukey's post-hoc test due to the lack of normality and non-equal variance in the dataset. Regression analyses were used to determine relationships between N water concentrations, Chl-*a*, % TOM and N fluxes. All statistics were performed using SigmaPlot 14.0.

3.3 Results

3.3.1 N Water Concentrations

NH_4^+ concentrations were highly variable across rivers (Table 3.1). Average concentrations at restored sites ranged from $36 \mu\text{g l}^{-1}$ to $731.7 \mu\text{g l}^{-1}$ and at unrestored sites from $28.0 \mu\text{g l}^{-1}$ to $290.5 \mu\text{g l}^{-1}$. However site-specific ranges were much greater (Table 3.1). Concentrations were significantly different among rivers ($F_{4,171} = 75.80$; $p < 0.001$) and significantly greater at restored reaches ($F_{4,171} = 28.26$; $p < 0.001$). There was also a significant interaction between river and restoration ($F_{1,171} = 18.65$; $p < 0.001$), although this was mainly due to the elevated concentrations at the Brent.

Average NO_3^- site concentrations at restored sites ranged from 9.6 mg l^{-1} to 23.7 mg l^{-1} whilst those at unrestored sites ranged from 9.6 mg l^{-1} to 26.4 mg l^{-1} (Table 3.1). NO_3^- concentrations differed significantly between rivers ($F_{4,170} = 282.94$; $p < 0.001$) but were not influenced by restoration ($F_{1,170} = 2.71$; $p = 0.10$). No significant interactions were found between rivers and restoration ($F_{1,170} = 2.34$; $p = 0.06$).

Table 3.1. A summary of ranges and averages of NH_4^+ ($\mu\text{g l}^{-1}$) and NO_3^- (mg l^{-1}) concentrations during the spring months of 2016. Values in parentheses denote one standard error. Significant differences between restored and unrestored reaches are in bold; difference among rivers are indicated by letter groupings.

Concentration	Ravensbourne		Pool		Wandle		Hogsmill		Brent	
	Rest.	Unrest.	Rest.	Unrest.	Rest.	Unrest.	Rest.	Unrest.	Rest.	Unrest.
NH_4^+ Average	146.3 ^a	151.0 ^a	141.3 ^{a,b}	115.0 ^{a,b}	36.0 ^c	28.0 ^c	79.5 ^{b,c}	56.5 ^{b,c}	731.7^d	290.5^d
	(33.7)	(32.2)	(24.0)	(11.8)	(6.1)	(4.8)	(7.4)	(5.4)	(84.3)	(21.8)
NH_4^+ Range	37.33-	38.00-	53.7-	53.5-	8.5-	11.5-	47.9-	31.1-	241.8-	202.0-
	438.8	406.8	536.8	266.9	103.5	103.2	146.3	106.2	1022	471.0
NO_3^- Average	12.4 ^a	12.5 ^a	13.0 ^b	12.6 ^b	23.7^{a,b}	26.4^{a,b}	22.7 ^{a,b}	23.3 ^{a,b}	9.6 ^{a,b}	9.55 ^{a,b}
	(0.7)	(0.7)	(0.5)	(0.5)	(0.7)	(0.3)	(0.6)	(0.3)	(0.67)	(0.7)
NO_3^- Range	6.6-	8.8-	7.7-	8.4-	16.5-	24.3-	14.3-	21.2-	7.3-	6.3-
	15.6	17.0	16.3	15.2	27.7	29.3	28.2	26.9	15.3	13.6

3.3.2 N Flux across the Sediment-Water Interface

Across the entire experiment both NH_4^+ and NO_3^- fluxes showed both uptake and regeneration and no constant patterns in magnitude or direction among these measurements (Table 3.2). Average NH_4^+ fluxes for 0-3 minutes across all rivers ranged from -8.9 to $3.4 \mu\text{g N m}^{-2} \text{sec}^{-1}$ and did not differ significantly between rivers ($F_{4,158} = 1.25$; $p = 0.29$) (Figure 3.3a; Table 3.2). There were no significant differences in 0-3 minutes NH_4^+ fluxes between restored and unrestored sites ($F_{1,158} = 0.02$; $p = 0.88$) (Figure 3.3b). NH_4^+ fluxes for 3-10 showed both uptake and regeneration (-7.1 to $7.5 \mu\text{g N m}^{-2} \text{sec}^{-1}$) and were significantly different ($F_{4,158} = 3.20$; $p = 0.015^*$) (Figure 3.3b; Table 3.2). However, restoration had no influence on 3-10-minute fluxes ($F_{1,158} = 0.42$; $p = 0.52$; Figure 3.3b).

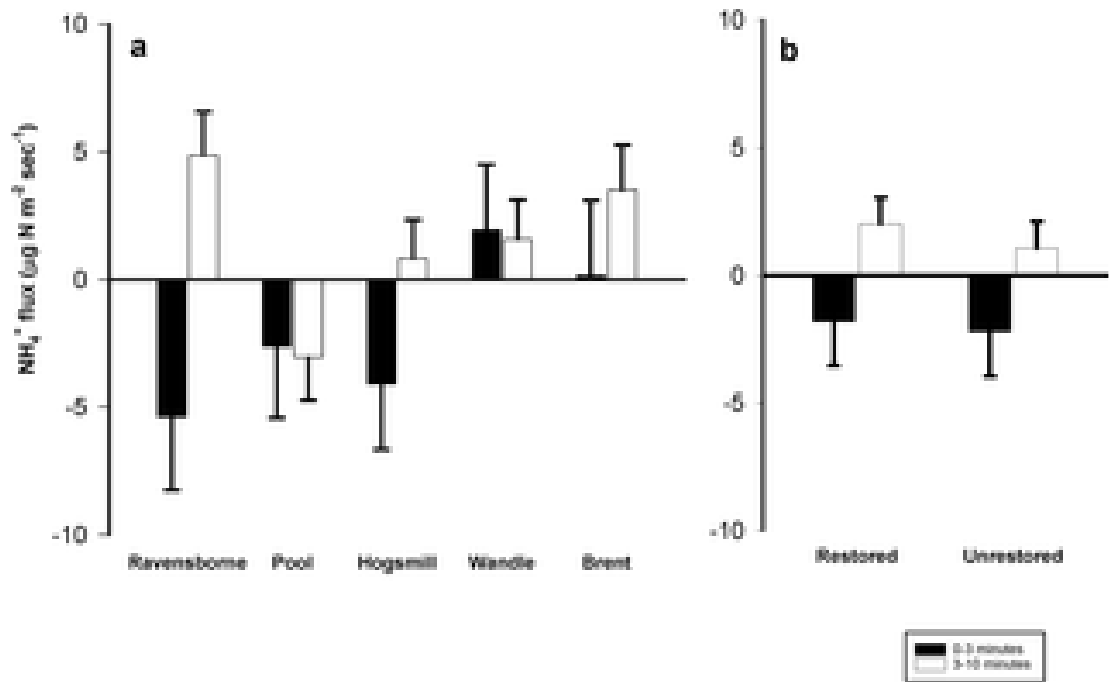


Figure 3.3. Average NH_4^+ fluxes ($\mu\text{g N m}^{-2} \text{sec}^{-1}$) among (a) the study rivers (restored and unrestored combined) and between (b) the combined restored and unrestored reaches from all London rivers. Columns represent average values ($n = 12-20$) + one standard error. Both physical disturbance ($T = 0-3$ minutes) and biogeochemical activity ($T = 3-10$ minutes) are presented in each panel. There was no significance different between river NH_4^+ fluxes over the 0-3 minutes period, nor between restored or unrestored reaches at both 0-3 and 3-10 minutes. Rivers with different letters show significant differences in fluxes over the 3-10 minutes. Positive flux values represent uptake/removal of nutrients from the water column and negative flux values represent release of nutrients from the sediment (regeneration).

NO_3^- fluxes for 0-3 minutes across all sites ranged from -33.6 to $97.8 \mu\text{g N m}^{-2} \text{sec}^{-1}$ (Table 3.2), with uptake in the restored sites and regeneration in the unrestored sites ($F_{1,158} = 6.14$; $p = 0.014^*$; Figure 3.4b; Table 3.2). However, there were no differences among rivers ($F_{4,158} = 1.1$; $p = 0.36$; Figure 3.4a; Table 3.2). Average NO_3^- fluxes for 3-10 minutes across all sites ranged from -14.4 to $16.0 \mu\text{g N m}^{-2} \text{sec}^{-1}$. No significant differences were found between restored and unrestored reaches ($F_{1,158} = 0.28$; $p = 0.60$; Figure 3.4b), or among the study rivers ($F_{4,158} = 2.38$; $p = 0.05$; Figure 3.4a; Table 3.2).

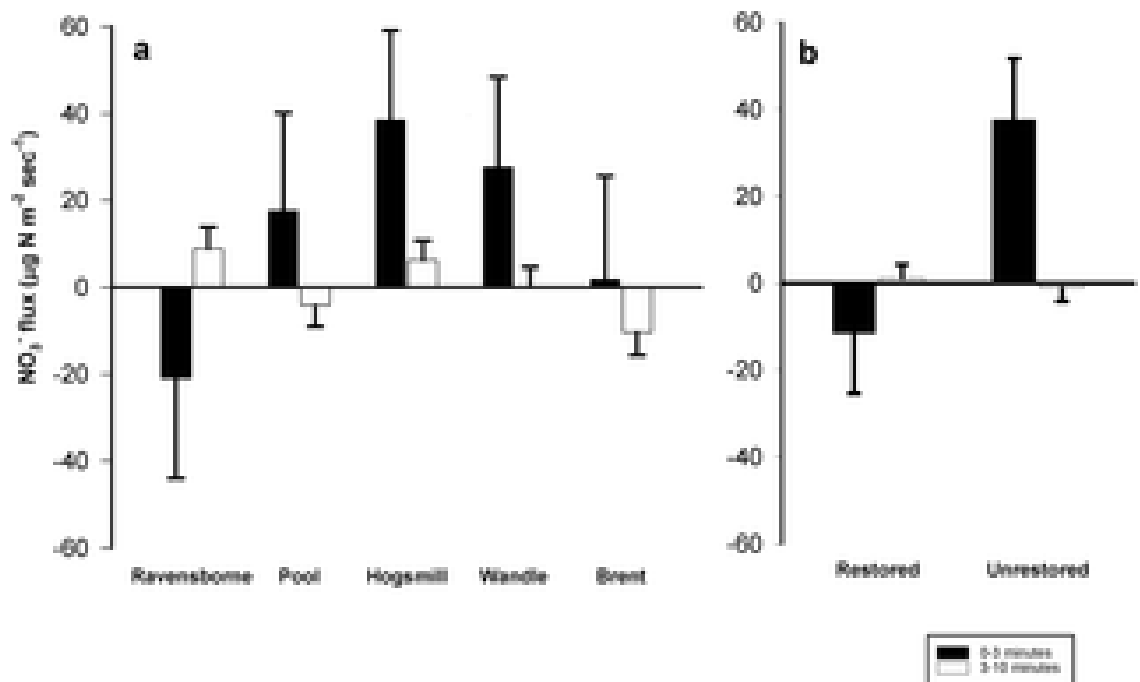


Figure 3.4. Average NO_3^- fluxes ($\mu\text{g N m}^{-2} \text{sec}^{-1}$) among a) the study rivers (restored and unrestored combined) and between (b) the combined restored and unrestored reaches from all London rivers. Columns represent average values ($n = 12-20$) + one standard error. Both physical disturbance ($T = 0-3$ minutes) and biogeochemical activity ($T = 3-10$ minutes) are presented in each panel. There was no significant difference in NO_3^- fluxes between rivers. However, there was a significant regeneration of NO_3^- from sediment in unrestored sites over the 0-3 minutes period, but no difference between fluxes at 3-10 minutes. Positive flux values represent uptake/removal of nutrients from the water column and negative flux values represent release of nutrients from the sediment (regeneration).

Table 3.2. A summary of N flux averages ($\mu\text{g m}^{-2} \text{sec}^{-1}$) of site-specific measurement ($n = 20$). Values in parentheses denote one standard error. Significant differences between restored and unrestored reaches are in bold; differences among rivers are indicated by letter groupings. Brown cells denote sediment uptake, blue cells denote regeneration. Overall, there were no constant patterns in the magnitude or direction of flux among all measurements.

Flux	Ravensbourne		Pool		Wandle		Hogsmill		Brent	
	Rest.	Unrest.	Rest.	Unrest.	Rest.	Unrest.	Rest.	Unrest.	Rest.	Unrest.
0-3 min NH_4^+ flux	-8.9	-1.9	1.0	-6.3	3.3	0.5	-2.9	-5.4	-1.7	2.0
	(5.9)	(4.7)	(2.1)	(5.4)	(1.3)	(1.6)	(2.8)	(3.2)	(6.7)	(4.0)
3-10 min NH_4^+ flux	2.2	7.5	1.0	-7.1	1.6	1.5	0.2	1.5	5.0	2.0
	(1.4)	(5.8)	(0.6)	(5.0)	(0.8)	(0.3)	(0.9)	(0.9)	(2.3)	(1.2)
0-3 min NO_3^- flux	-32.6	-9.3	28.4	6.7	0.3	54.6	-20.9	97.7	-33.6	37.0
	(24.1)	(21.7)	(28.1)	(28.8)	(34.8)	(40.8)	(19.2)	(44.8)	(19.0)	(22.4)
3-10 min NO_3^- flux	16.0	1.6	-13.5	5.2	-4.8	4.9	14.8	-2.6	-6.4	-14.4
	(5.6)	(5.6)	(6.8)	(4.9)	(7.1)	(9.1)	(8.0)	(5.1)	(3.2)	(7.2)

3.3.3 Relationship Between Sediment Grain Size, Chl-*a*, % TOM and Flux

Sediment grain size among all sampling locations varied little and was predominantly sand (Table 3.3). Average Chl-*a* concentrations at restored sites ranged from 0.3 to 1.9 $\mu\text{g g}^{-1}$, while those at unrestored sites ranged from 0.4 to 0.7 $\mu\text{g g}^{-1}$ (Table 3.3). Chl-*a* differed among rivers ($F_{4,148} = 2.95$; $p = 0.02$), with the Wandle differing from the Hogsmill and Pool. There were also significant differences between restored and unrestored reaches at the Wandle ($p = 0.003$) (Table 3.3). However, restoration did not have an overall effect on Chl-*a* concentrations between restored and unrestored reaches ($F_{1,148} = 2.52$; $p = 0.12$). Average % TOM ranged from 18.54 to 30.83 across restored and unrestored reaches (Table 3.3), but did not differ significantly among rivers ($F_{4,158} = 2.22$; $p = 0.070$). %TOM was also significantly higher at the restored Wandle site ($p = 0.008^{**}$), but this was not consistent across all reaches ($F_{4,158} = 0.80$; $p = 0.37$). Across all of the regressions, there were no significant relationships found between N water concentrations, % TOM and Chl-*a* to either NH_4^+ and NO_3^- flux associated with disturbance or biogeochemical activity (i.e. $R^2 < 0.09$; $p > 0.05$ among all relationships).

Table 3.3. A summary of the average ($n = 12-20$) sediment grain size, Chl-*a* and % total organic matter. Values in parentheses denote one standard error. Differences between rivers are indicated by letter groupings.

Parameter	Ravensbourne		Pool		Wandle		Hogsmill		Brent	
	Rest.	Unrest.	Rest.	Unrest.	Rest.	Unrest.	Rest.	Unrest.	Rest.	Unrest.
Sediment grain size	92.57 (0.4)	94.29 (0.6)	96.41 (0.2)	95.76 (0.6)	90.17 (0.6)	97.37 (0.1)	91.78 (1)	94.27 (0.1)	95.52 (0.3)	97.80 (0.2)
Chl-<i>a</i>	0.57 ^{a,b,c} (0.1)	0.46 ^{a,b,c} (0.1)	0.33 ^{a,b} (0.1)	0.57 ^{a,b} #(0.1)	1.92^c (0.09)	0.69^c (0.2)	0.57 ^{b,a} (0.1)	0.44 ^{b,a} (0.1)	0.76 ^{a,b,c} (0.2)	0.39 ^{a,b,c} (0.1)
TOM	20.64 ^a (1.7)	19.38 ^a (2.2)	18.54^a (2.2)	26.57^a (2.8)	30.83^a (4.5)	21.14^a (2.4)	27.06 ^a (2.6)	27.02 ^a (1.5)	28.38 ^a (3.8)	23.52 ^a (2.7)

3.4 Discussion

The overall aim of this study was to understand the effects of restoration on nitrogen processing across the sediment-water interface in restored and unrestored urban streams. Results from this study indicate that restoration had no consistent overall effect on NH_4^+ and NO_3^- uptake or regeneration rates (Fig 3.3 and 3.4; Table 3.2). These findings meet the first two aims of this study and reject the first hypothesis which postulated that restoration would facilitate increased nitrogen uptake due to assimilation. These uptake values are in line with those reported across a range of stream types for NH_4^+ flux (Bernot and Dodds, 2005; Tank *et al.*, 2018) and NO_3^- flux (Bernot and Dodds, 2005; Grimm *et al.*, 2005; Hall *et al.*, 2009). This is perhaps not surprising given the highly urban nature of London rivers where nutrient loading and sediment N saturation are likely to be offsetting any N removal associated with restoration (Walsh *et al.*, 2005; Mulholland *et al.*, 2008). Furthermore, these results are similar to those seen in urban systems (Bernot and Dodds, 2005; Reisinger *et al.*, 2016; O'Brien *et al.*, 2007) and restored sites (Klocker *et al.*, 2009; Suddith *et al.*, 2011). It is possible that at restored reaches of the Ravensbourne, Pool, Wandle and Hogsmill, in-stream berms and cobbles which have been deployed to renaturalise flows may have simultaneously stimulated sediment deposition and facilitated N assimilation. In terms of average differences in uptake and regeneration between 0-3 and 3-10 minutes, there was no consistent pattern between the two (hypotheses 1-3). Physical disturbances increased uptake of NH_4^+ between 0 and 3 minutes, however after 3 minutes, NH_4^+ was released back into the water column.

Given the extent of N loading among the study rivers, coupled with the varying timescales over which ecological and chemical indices respond to restoration, it is not surprising that results were equivocal. This is further supported by previous studies which have found variable responses of restoration on N dynamics (Roni *et al.*, 2008; Smith and Chadwick, Filoso and Palmer, 2011; 2014). In this study there was insufficient evidence to suggest

that restoration is leading to improvements in either water quality (Table 3.1) or N flux (Fig 3.3 and 3.4, Table 3.2). Even for projects where ecological characteristics may positively respond to reach-scale restoration, it is likely that poor water quality throughout catchments may impinge upon any significant improvements. However, whilst this lack of “restoration effect” was consistent, a caveat is that they come from five unique streams in urban London, with data collected at a patch scale.

Across all the study sites, restoration practices did not lead to significant reductions in NH_4^+ or NO_3^- concentrations (Table 3.1). NH_4^+ concentrations varied widely across sites, aligning with previously reported values observed in London tributaries (EEA, 2017; Pecorelli, 2017). This highlights the heavily impacted nature of London rivers upon which multiple stressors are acting. In contrast, NO_3^- concentrations differed significantly from previous studies, highlighting a >50% rise in concentrations greater than 20 mg l^{-1} at the Wandle and Hogsmill and a concentration decrease of a similar magnitude at the Brent (Table 3.1). These concentrations are comparable to previously reported values within the Thames catchment (Pennino *et al.*, 2014; Halliday *et al.*, 2015; Van Biervliet *et al.*, 2017) but often lower than other urban rivers of Europe which can exceed 100 mg l^{-1} (Floury *et al.*, 2013; Minaudo *et al.*, 2015). Higher concentrations of NO_3^- versus NH_4^+ were observed across all sites, which may be attributed to nitrification processes occurring in-stream and uptake distances that are shorter for NH_4^+ than NO_3^- (Peterson *et al.*, 2001; Grimm *et al.*, 2005; Kaushal *et al.*, 2008). Previous links have been made between inorganic N inputs in headwater streams and rapid N removal which highlight the potential for removal or transformation across small temporal and spatial scales (Alexander *et al.*, 2000; Peterson *et al.*, 2001). However, this is not the case in London streams, probably due to N sediment saturation and continuous pollution loading (Paul and Meyer, 2001; Walsh *et al.*, 2005).

Initially, it was surprising that overlying NH_4^+ and NO_3^- concentrations did not correspond with uptake or regeneration fluxes. Several studies have reported positive relationships between N concentrations and uptake in urban streams resulting from restoration activities (Grimm *et al.*, 2005; Roberts *et al.*, 2007; Hall *et al.*, 2009; Hines and Hershey, 2011; Pennino *et al.*, 2014). Other studies have highlighted the role of biogeochemical transformations in triggering NO_3^- reduction to NH_4^+ and N_2 in anaerobic sediments (Clavero *et al.*, 2000). The highly urban nature of these study streams combined with potential N removal and transformations (ammonification, nitrification and denitrification) across the sediment-water interface may explain these differences. This is supported by previous studies which have identified that urban cover >20% can hinder stream responses to restoration (Walsh *et al.*, 2005; Chadwick *et al.*, 2006). Percentage urban cover at sites used for this study far exceed these values, ranging from 47-69% (Table 2.1).

There were no relationships observed between Chl-*a*, organic matter, sediment size and N flux, as postulated in the final aim of this study. This is likely due to increases in N concentrations reducing the capacity of streams to retain and transform N inputs, leading to a reduction in biotic uptake and denitrification (Peterson *et al.*, 2001; Mulholland *et al.*, 2008; Kaushal and Belt, 2012). This supports equivocal relationships observed between Chl-*a* and N flux, which differs from other studies linking Chl-*a* to N concentrations, % TOM and suspended sediments (Neal *et al.*, 2006b). This meets the third objective of this study. Significant NO_3^- uptake rates were recorded at the Ravensbourne, Hogsmill and Brent following physical disturbances (e.g. 0-3 treatment). This may be attributed to N dynamic ‘hotspots’ for NO_3^- uptake and assimilation following disturbances. However, no significant relationship was observed for the biogeochemical flux, thus it is difficult to determine any restoration success related to N

dynamics. Biogeochemical processing of flux between N dynamic ‘hotspots’ and ambient water warrants further research, specifically looking at nutrient uptake limitations and relationships between N supply and biological demand (Covino *et al.*, 2018).

This method using N flux assays in small chambers focuses on processes which occur at the sediment-water interface. However, making comparisons with other research projects reporting spiralling is not straightforward. Therefore, methods for adapting this approach to allow for upscaling to evaluate impacts to downstream systems needs further development. Despite these issues, these results do provide strong evidence to show that current restoration practices are unlikely to support predictable changes in N dynamics without greater understanding of site-specific factors which affect disturbance and biogeochemical-associated fluxes (Klocker *et al.*, 2009; Sivorichi *et al.*, 2011).

3.4.1 Future Management Recommendations

Reach-scale restoration did not influence N flux in this study. This should not necessarily be perceived as a restoration failure, but an opportunity to examine restoration responses across different spatial and temporal scales. Given the small size of restored reaches within this study and the urban catchments which experience a myriad of multiple stressors (Walsh *et al.*, 2005; Meyer *et al.*, 2005; Chadwick *et al.*, 2006; McGrane, 2016; Vietz *et al.*, 2016), it is perhaps not surprising that no significant benefits were accrued. In combination with the delayed response of pollutants to restoration, this highlights the need for larger scale restoration studies to be undertaken over prolonged timescales. Whilst many projects examine the fate of accumulated N in middle and downstream reaches (Bernot and Dodds, 2005; Mulholland *et al.*, 2008; Alexander *et al.*, 2009), few focus on targeting N inputs in headwater streams (Peterson *et al.*, 2001; Simon *et al.*, 2010). Headwater reaches are highly susceptible to nutrient loading from urban land, therefore restoration could provide widespread potential to mitigate against

eutrophication associated with N loading (Clavero *et al.*, 2000; Craig *et al.*, 2008; Wu and Yang, 2012). Selecting restoration sites in headwaters based on optimal area, size, discharge and velocity dimensions can positively influence N uptake metrics (Johnson *et al.*, 2015). This will help to create a buffer for downstream environments where an increasing urban gradient is likely to reduce N removal capacity.

The range of restoration practices applied to these study sites did not produce consistent results, therefore additional restoration practices could potentially improve the condition of these urban rivers. For example, Sustainable Urban Drainage Systems (Zhou, 2014) have the potential to remove N through wetlands, swales and attenuation ponds across sensitive catchment areas. Stream daylighting is also increasingly being adopted as a restoration strategy to increase hyporheic exchange and eliminate excess N in the presence of bioavailable carbon (Neale and Moffett, 2016). Integrating vegetative structures can help to restore natural flow regime resulting from channelisation whilst combatting problems associated with thermal stress (Paul and Meyer, 2001; Walsh *et al.*, 2005; Kim *et al.*, 2008). Future restoration projects should seek to determine how habitat alterations and hydrological regime can stimulate N uptake whilst building resilience to disturbance events (Wohl *et al.*, 2015). Irrespective of these management options, rivers in London and other cities still have a legacy of widespread misconceptions which are contributing to significant amounts of effluent entering these urban rivers.

3.5 Conclusions

This study sought to determine whether river restoration activities could influence N dynamics within degraded rivers in London. This small-scale approach highlighted the dynamic nature of N processing occurring within urban river reaches. Results highlighted that NH_4^+ concentrations were significantly higher at restored sites than at unrestored sites, whilst NO_3^- concentrations did not differ between reaches. Overall, restoration did

not significantly alter NH_4^+ or NO_3^- fluxes. This suggests that a synergy of geomorphic and biogeochemical processes (including natural and artificial stream morphology, stream bed characteristics, availability of nutrients and temperature) are also likely to be influencing N processing, necessitating further study.

There is a critical need to better understand the mechanisms controlling the inputs, processing and transformations of NH_4^+ and NO_3^- in urban river systems. This is particularly true for the highly urbanised systems found in megacities like London which far exceed impervious cover characteristics of other cities. Future research should focus on incorporating combined on-site outfall identification work and tracer studies to determine the source, saturation concentrations and fate of N. Supporting studies should examine other environmental variables that may be influencing flux dynamics. Sediment-water nutrient interactions have historically been overlooked in restoration studies in favour of aesthetic, hydrological and biological improvements. If the overall aim of river restoration is to improve ecosystem function, these factors should be considered as interacting components to maximise the chance of ecosystem recovery and build resilience to future perturbations.

Chapter 4: A Comparative Study of Organic Matter

Assays to Examine Decomposition in Urban Streams

Abstract

Decomposition is the catabolism (breakdown) of organic matter into its organic constituents including carbon dioxide, inorganic nitrogen and phosphorus. This is an essential process for maintaining the structural and functional integrity of freshwater and terrestrial ecosystems through leaching of soluble compounds and microbial activity (fungal and bacterial), including conditioning and bacterial feeding. Several processes influence decomposition (directly or indirectly) including physical abrasion, temperature, invertebrate shredding and microbial activity. Leaf litter assays are a common method used to quantify decomposition rates, however they incur several shortcomings such as variations amongst litter species, leaf quality and environmental variables (e.g. sunlight). Increasingly, alternative organic matter materials such as tea bags and cotton strips are being used as leaf litter surrogates due to their widespread uniformity and ease of use. The primary aim of this study was to *compare decomposition rates across restored and unrestored reaches using multiple assay methods*. The objectives were (1) To determine whether decomposition rates vary between restored and unrestored sites using coarse and fine mesh bags to decipher macroinvertebrate and microbial contributions to breakdown, and (2) To evaluate the effectiveness of different assay techniques including leaf litter, cotton strips and tea bags to measure decomposition. It was hypothesised that decomposition would differ significantly between restored and unrestored sites, owing to processes including physical abrasion, nutrient processing and biological activity. It was also hypothesised that cellulose cotton strips would provide a preferential tool to measure decomposition over leaf litter or tea bags due to their uniformity and ease of application. It was also expected that flow velocities would influence breakdown. Across five paired

restored and unrestored reaches of River Thames tributaries, London Plane *Platanus x hispanica* leaf litter (coarse and fine mess bags), rooibos tea bags and 96% cellulose cotton strips were incubated for 28 days to examine rates of decomposition. Rates were measured as units of mass loss (per degree day (k_{dd}) for leaves and tea bags, and loss of tensile-strength and % loss k_{dd} for cotton strips. Stream velocity (ms s^{-1}), depth and water chemistry (temperature, conductivity, pH, total organic carbon (TOC) and nitrate (NO_3^-) were measured as possible causes of breakdown. Decomposition rates for coarse leaf litter ($0.002 k_{dd}$), tea bags ($0.001\text{-}0.002 k_{dd}$) and cotton strips ($0.23\text{-}0.36\%$ loss k_{dd}) did not differ between restored and unrestored sites. However, significant differences were recorded for breakdown for fine bags across reaches ($F_{1,22} = 7.11$; $p = 0.01$), due to elevated rates of decay at the Brent restored reach ($p = 0.009$). Temperature was considered to be an important factor influencing breakdown among sites, however physicochemical variables assessed in this study were not considered to be a significant factor driving breakdown. Cotton strips decomposed more rapidly than leaf litter and tea bags, highlighting their sensitivity to environmental change. Future studies would benefit from adopting this uniform approach to increase the likelihood of detecting the main factors influencing decomposition in urban streams.

4.1 Introduction

4.1.1 Organic Matter Processing

Leaves are important mediators of ecosystem function, subject to in-stream processes such as physical fragmentation, shredding by macroinvertebrates, metabolic breakdown by microorganisms and nutrient cycling (Bocock and Gilbert, 1957; Hieber and Gessner, 2002; Tank *et al.*, 2010). Sources of stream organic matter include leaf litter coarse particulate organic matter (CPOM), riparian vegetation and dissolved organic matter (DOM). Stream ecologists typically classify organic matter by size into coarse particulate organic matter (CPOM >1 mm), fine particulate organic matter (FPOM ~0.45 µm) and dissolved organic matter (DOM < ~0.45 µm) (Figure 4.1). Benthic residence time of organic matter can range from days (e.g. labile sugars) to years (e.g. woody debris) and is largely related to the size of different organic matter fractions (particulate, dissolved or coarse) and substrate type (Aumen *et al.*, 1983). Different organic matter fractions are typically retained at different locations within the river habitat. For example, CPOM is retained close to the point of entry in the presence of trapping features such as debris dams. In turn, woody debris can decrease stream velocity and contribute to particulate organic matter (POM) retention. DOM processing in urban catchments is poorly understood, however urbanisation can modify labile fluxes by increasing microbial production and respiration in headwaters (Khamis *et al.*, 2017).

4.1.2 Causes of Decomposition

Abiotic factors, substrate chemistry and decomposers all influence the breakdown of leaf litter. Shredders are aquatic invertebrates that consume coarse particulate organic matter (CPOM) and provide fine particulate organic matter (FPOM) for other organisms through feeding and excretion (Cummins *et al.*, 1989). Increases in density of generalist invertebrate species such as *Asellus aquaticus* and *Gammarus pulex* are often correlated with increased rates of breakdown (Tank *et al.*, 2010). Shredders favour leaves which

have undergone conditioning by microbes (due to their softening mechanism) and contain a higher amount of nutrients (particularly nitrates) and secondary compounds (Irons *et al.*, 1988).

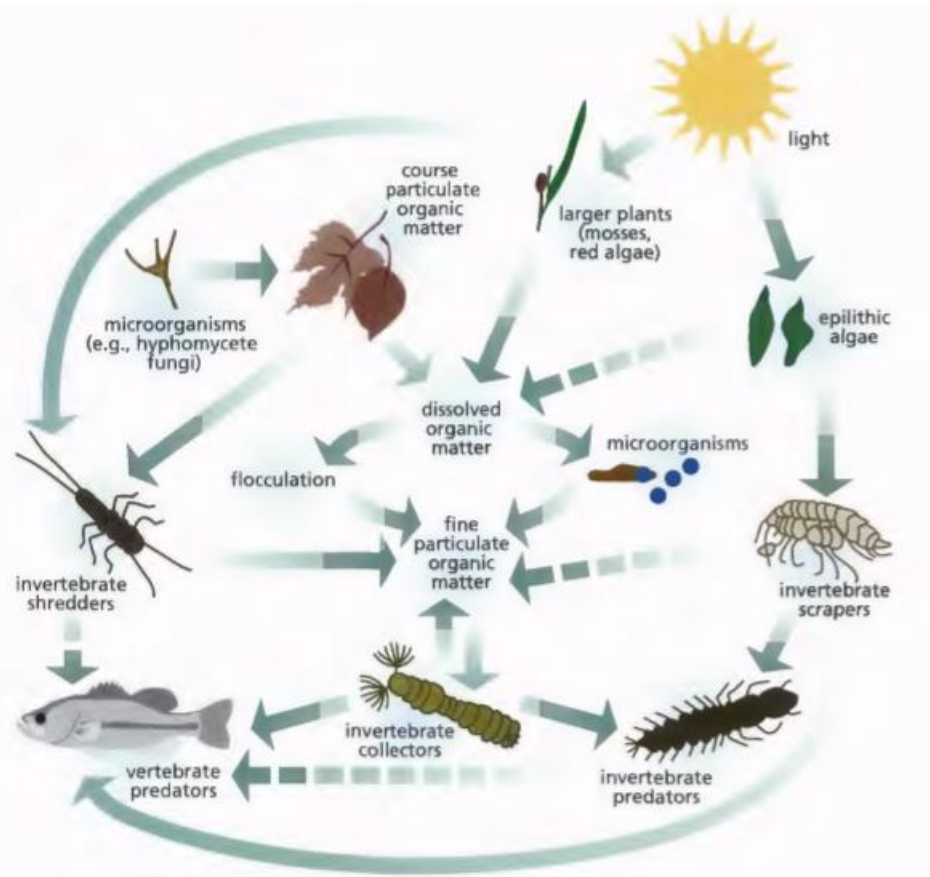


Figure 4.1. Stream biota, showing food relationships typically found in streams. Illustration showing organic matter components (coarse, fine and dissolved material) and interacting ecology. Image obtained from the 'Stream Corridor Restoration: Principles, processes and Practices (1998)', by the Federal Interagency Stream Restoration Working Group.

Nutrients can lead to both the proliferation and reduction in activity of microorganisms breaking down leaf litter (Pascoal *et al.*, 2005). In polluted streams, microorganisms present in submerged leaf litter obtain nitrogen and phosphorus from water, microbial respiration, biomass and sporation (Gulis and Superkropp, 2003; Pascoal *et al.*, 2005; Young *et al.*, 2008). In turn, this increases decomposition and palatable detrital availability for invertebrates (Gulis and Suberkropp, 2003). In contrast, other studies have documented graded responses to hyphomycete (fungal) activity in highly polluted waters (Gulis *et al.*, 2004).

“Compensatory effects” can also occur, whereby increases in pollution alongside decreases in velocity and oxygen can influence the survival of invertebrates and microbes essential for decomposition (Pascoal and Cassio, 2004). Heterotrophic microorganisms (fungi and bacteria) condition leaves through mineralisation, rendering them more palatable for shredding by macroinvertebrates which gain nutritional value from both leaf litter and microbial biofilms (Superkropp, 1998; Franken *et al.*, 2005). However, in streams subject to significant pollution loads (e.g. mine effluent), reductions in breakdown rates have been associated with increases in heavy metal waste and reduced pH values, which can inhibit microbial growth and amphipod activity (Hogsden and Harding, 2013).

Water temperature is an important control on organic matter breakdown due to its effect on stream metabolism (Abelho *et al.*, 2005). Studies examining the role of temperature on breakdown have noted increased levels of shredding in upstream environments which are characterised by cool, leaf filled tributaries relative to area size (Baldy *et al.*, 1995). Furthermore, these environments provide isolated habitats to support Plecoptera (stonefly) and Trichoptera (caddisfly) populations.

Yule *et al.* (2015) conducted an experiment to assess the effects of urbanisation on decomposition. They predicted that urbanisation would lead to a reduction in leaf litter decomposition mediated by shredders and an increase in decomposition driven by microbes. A range of parameters were examined including leaf litter colonisation by macroinvertebrates (including richness), colonisation of tiles by algae and leaf litter breakdown. The highest rates of decomposition for both fine and coarse mesh bags occurred at the most severely impacted site, which was also characterised by significantly higher water temperatures. Current velocity and stream discharge can further increase leaf

fragmentation, a process commonly associated with urban streams (Paul and Meyer, 2001; Chadwick *et al.*, 2006).

4.1.3 The Utility of Decomposition Assays

The leaf litter assay, used widely by stream ecologists and soil scientists, has improved the knowledge of decomposition processes (Webster and Benfield, 1986). This method, used to compare breakdown rates in response to multiple environmental factors, is applied by incubating leaves in the field and later collecting them to determine mass loss over time (Zeglin, 2015). Leaf litter assays have been credited with being cost-effective and easily applicable across different spatial and temporal environments. An effective approach used to understand macroinvertebrate and microbial contributions to breakdown is through the use of coarse and fine leaf litter assays (Smith and Chadwick, 2014). However, this approach has several shortcomings (Webster and Benfield, 1986). Decomposition rates vary between tree species, leaf quality and in response to herbivore-induced plant defences. Other confounding factors include intra-species variations and sunlight exposure which can alter nutrient contents and palatability for shredders and microbes (Sariyildiz and Anderson, 2003). The tendency of invertebrates to consume leaves containing a greater amount of nutrients (e.g. nitrogen), whilst rejecting leaves of increased toughness and those containing chemicals (e.g. polyphenolics, influencing digestion and a feeding deterrent), can significantly influence breakdown rates (Irons *et al.*, 1988). These synergistic factors are particularly difficult to untangle in urban streams which may be subject to a multitude of compensatory factors. Finally, variations in bag size, shape and mesh size prior to deployment can also influence accessibility for shredders.

An alternative to the leaf litter assay are commercially available tea bags due to their ease of use, low-cost and uniformity (Didion *et al.*, 2016). Tea bag assays provide the benefit

of high-resolution measurements for decomposition, which can be modelled and scaled up to an ecosystem level to determine decomposition rates more accurately (Didion *et al.*, 2016). Whilst the teabag assay has been increasingly used in terrestrial studies including soil science research (Keuskamp *et al.*, 2013; Djukic *et al.*, 2018), knowledge of this assay in river environments is limited. Previous studies which examined decomposition of rooibos tea bags recorded no effects of land-use on early stage litter decomposition (Djukic *et al.*, 2018). Nevertheless, multiple environmental drivers were considered to influence mass loss, thus providing an opportunity to cross-compare different methodologies (Djukic *et al.*, 2018). Drawbacks associated with using the tea bag approach include varying decomposition rates in response to species types; fragility of the tea bag exterior and sedimentation within the tea bag (Keuskamp *et al.*, 2013).

Cotton strips have been applied in soil science studies and can act as a leaf litter surrogate in stream decomposition studies (Van Gestel *et al.*, 2003). Cotton strips are free of nutrients (N and P), less prone to fragmentation and provide a uniform substrate (Wallace *et al.*, 1997, Benfield *et al.*, 2001). Cotton strips are predominantly comprised of cellulose, a preferential substrate for leaf-colonising fungi and bacteria and a food source for leaf-shredding stream invertebrates. As with leaf litter, cotton strip decomposition can be assessed through measurements of mass loss. However, decomposition is more commonly measured over time as loss of tensile-strength (lbs) (Tiegs *et al.*, 2007; Tiegs *et al.*, 2013) and can be compared in relative terms to leaf litter decomposition. The standardisation of cotton strips is a major benefit when examining changes through times and space. Additionally, tensile-strength loss tends to occur more rapidly than that of leaf litter due to its favourability as a carbon source for macroinvertebrates and microbes (Tiegs *et al.*, 2007). The inconspicuous nature of cotton strips, combined with the shorter

incubation time needed for mass loss to occur, makes them less prone to loss or fragmentation from physical abrasion (Newman *et al.*, 2001).

The overall aim of this study was to compare decomposition rates across restored and unrestored reaches using multiple assay methods. The key objectives were **(1)** To determine whether decomposition rates vary between restored and unrestored sites using coarse and fine mesh bags to decipher macroinvertebrate and microbial contributions to breakdown and **(2)** To evaluate the effectiveness of different assay techniques including leaf litter, cotton strips and tea bags to measure decomposition. It was hypothesised that **(1)** decomposition would differ significantly between restored and unrestored sites, owing to processes including physical abrasion, nutrient processing and biological activity, and **(2)** cellulose cotton strips would provide a preferential tool to measure decomposition over leaf litter or tea bags to their uniformity and ease of application.

4.2 Materials and Methods

4.2.1 Assay Preparation

4.2.1.1 Leaf Litter

The leaf litter assay method was adopted from Smith and Chadwick (2014) who previously examined decomposition in urban streams of London. Freshly fallen London plane *Platanus x hispanica* leaves were collected in December 2017 from the Victoria Embankment, London. Plane trees are a non-native species that account for over 50% of trees in London and decompose quickly. Coarse mesh bags (mesh size approximately 5 mm x 10 mm) (Figure 4.2a) were selected and prepared for this study to enable access of invertebrates (which are important for shredding), during deployment. Bags were constructed from large nylon bags (Drawstring Raschel net sacks) to sizes of approximately 17 cm x 22 cm to ensure that leaves were comfortably contained inside without crushing. Leaves were air-dried for more than two weeks, separated from petioles

and weighed to the nearest five grams (± 0.1 g). Leaf edges were carefully trimmed where weight requirements were not met. Fine mesh bags (mesh size approximately 1 mm x 1 mm) (Figure 4.2b) were selected for this study to prohibit invertebrate access and attribute decomposition to possible breakdown by microbes. Fine mesh bags were cut from universal insect nets to smaller dimensions of approximately 10 cm x 5 cm, folded and stapled at the edges to enclose leaves. Leaves were air-dried for more than two weeks and weighed to the nearest three grams (± 0.1 g). Petioles were removed. Similarly, to coarse bags, leaves were trimmed at the edges where they did not meet the weight requirements to fit into bags. Prior to bagging, leaves were wetted in deionised water to soften and limit fragmentation. Fifteen additional coarse and fine breakage bags and tea bags were weighed and bagged to account for handling losses.

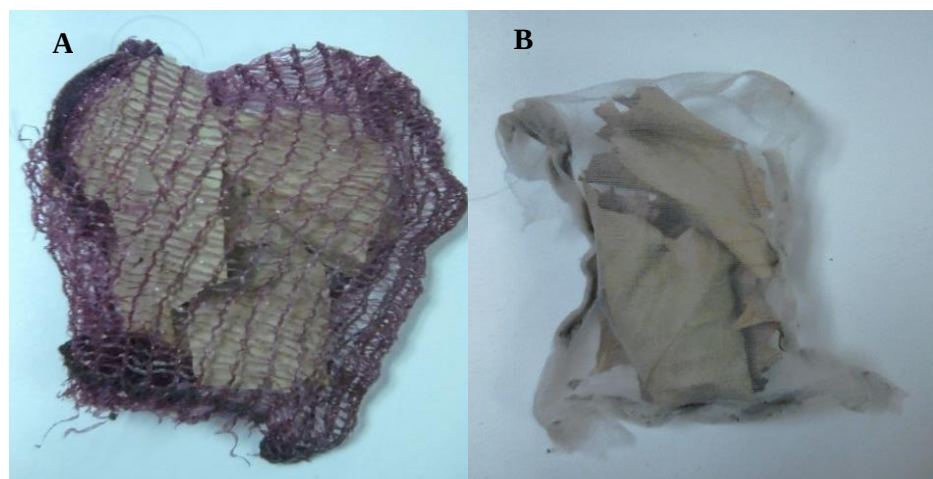


Figure 4.2. Mesh bags deployed at sites for **(a)** Coarse mesh bags (mesh size approximately 5 mm x 10 mm) containing five grams (± 0.1 g) of leaves and **(b)** Fine mesh bags containing three grams (± 0.1 g) of leaves.

4.2.1.2 Tea Bags

The tea bag assay was modified from methods by Keuskamp et al. (2013) who carried out tea bag decomposition assessments within terrestrial environments. One hundred and fifty tetrahedron-shaped synthetic rooibos tea bags (Figure 4.3) (Char Teas, containing South African shrub *Aspalathus Linearis*, approximately 1.3 g) were air dried, weighed

(to the nearest 0.0001 g) and inserted into coarse litter bags prior to deployment. Bags consisted of rooibos and were supplemented with natural flavouring. Mesh sizes of 0.3 mm allowed microorganisms to enter the bags but excluded macrofauna. Tea bag dimensions were approximately 6 cm x 6 cm.



Figure 4.3. *Synthetic rooibos tea bags (Keuskamp et al., 2013).*

4.2.1.3 Cotton Strips

The cotton strip assay followed procedures outlined by Tiegs et al. (2013). Most recent studies in the US have substituted the standardised Shirley Burial Test Fabric which has been used in decomposition studies for decades for a material conventionally used as artists' fabric (Tiegs *et al.*, 2013; Figure 4.4). Strips were prepared externally from bolts of Fredrix-brand unprimed 12-oz, heavy weight cotton fabric (style #548). Using a rotary cutter (OLFA brand, RTY-2/DX), strips were cut from a master cloth measured to dimensions of 25 mm width, 1.37 m length (precision 1-2 mm). Strips were made from 100% unbleached cotton and 96% cellulose with twenty-seven warp threads fringed by 3 mm of fray. Prior to deployment, strips were air dried, cut to approximately 9 cm, weighed (~0.8 g) and sorted into separate coarse nylon bags (15 per site). Bags were coded according to retrieval times (five replicates to be collected after 7 (A), 14 (B) and 28 days (C) (Figure 4.5).

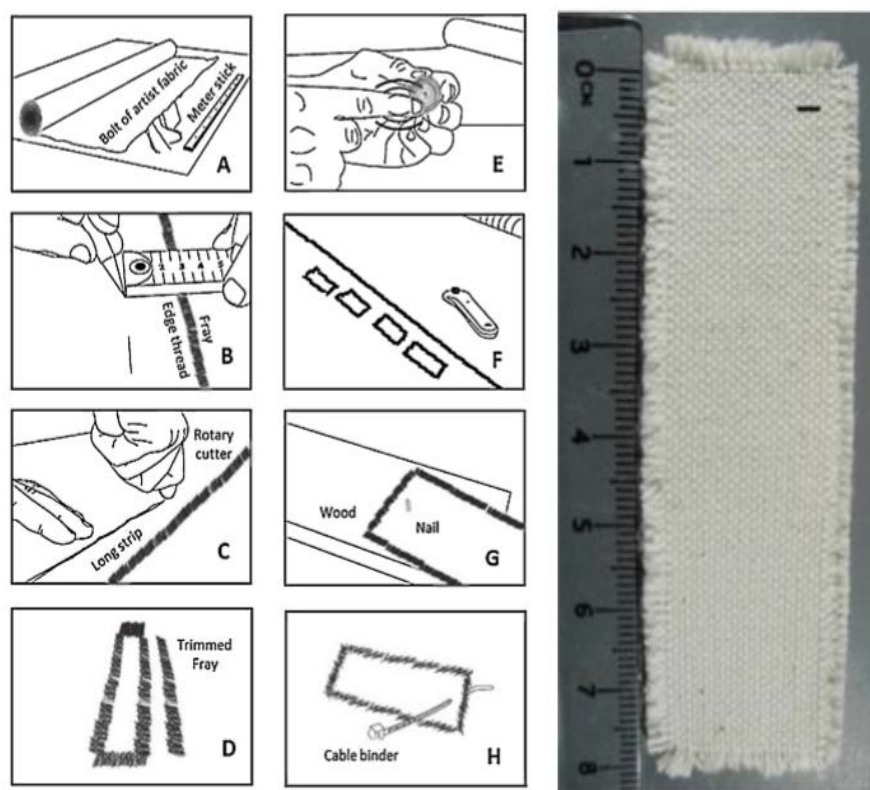


Figure 4.4. Construction of cotton strips from artists' fabric. (A) A partly unrolled bolt of the fabric. (B) 25 mm markings made inwards from the edge thread (not the edge of the fray). (C) Cutting along the edge of the cloth using a rotary cutter, ensuring the strip is 27 threads wide. (D) Trimmed fray so that only 3 mm of fray is remaining along the edge of the strip (E) Rolled strip to ensure the strip is facing outwards. (F) Strip is unrolled and cut into 8.0 cm long strips (G) A small incision being made using a thin nail that protrudes from a piece of wood which is between two threads at approximately 7 mm from one end of the strip. (Tiegs et al, 2013).

4.2.2 Field Methods

4.2.1.1 Physicochemical Measurements

At each site, 15 depth and velocity measurements were taken using a Valeport 801 Electro-Magnetic Open Channel Current Meter. To characterise physicochemical conditions, five spot samples were taken the same day for pH, conductivity ($\mu\text{S cm}^{-1}$), dissolved oxygen (ppm) and temperature ($^{\circ}\text{C}$) using a Hanna multiparameter water quality meter (HI98194). Five additional ambient water samples were obtained, filtered and stored in a cool bag for measurements of chloride, total organic carbon (TOC) and nitrate (NO_3^-).

4.2.1.2 Mixed-Assay Deployment

Mixed decomposition assays were deployed in mid-January 2018. Each site contained three sets (A, B and C) for collection after 7 days, 14 day and 28 days, respectively (Figure 4.5). Sets A and B included five coarse mesh bags, each containing one tea bag and one cotton strip (no leaf litter) and Set C (including five coarse mesh bags, each containing one tea bag, one cotton strip and approximately 5 g of leaf litter, and five fine mesh bags, each containing approximately 3 g of leaves). 5 repeats assays were made for each of these assays (Figure 4.5). Bags were carefully secured in-stream using tent pegs and cable wire at random locations across the 25 m reach. The decision of where to place bags was guided by pilot studies which detected assay losses due to high flows, rapid decomposition, sedimentation or disengagement from the stream bed. Bags were placed in-stream to reflect the range of reach conditions. This meant that that they were sometimes placed closer to the channel bank. 15 fine and coarse mesh breakage bags and tea bags were also carried into the field to account for handling losses.

After 7 days, Set A (five coarse bags containing cotton strips and tea bags) were removed. Cotton strips were gently wiped, soaked for 30 seconds in 70% ethanol to halt microbial activity, carefully placed in individual plastic bags and returned to the laboratory (Correll *et al.*, 1997). Tea bags were carefully contained in bags. The same method was applied to Set B after 14 days. After 28 days, Set C was recovered; both tea bags and cotton strips were treated in the same way as Sets A and B. Fine and coarse mesh bags containing leaves were retrieved and carefully stored. Leaves were only examined in this study after 28 days due to analytical time constraints. Timescales for deployment of leaves were modified based on previous studies of these streams (Smith and Chadwick, 2014). Cotton strip incubation times were selected based on previous studies which have indicated that

tensile-strength loss greater than 50% can provide an accurate representation of environmental conditions (Tiegs *et al.*, 2013).

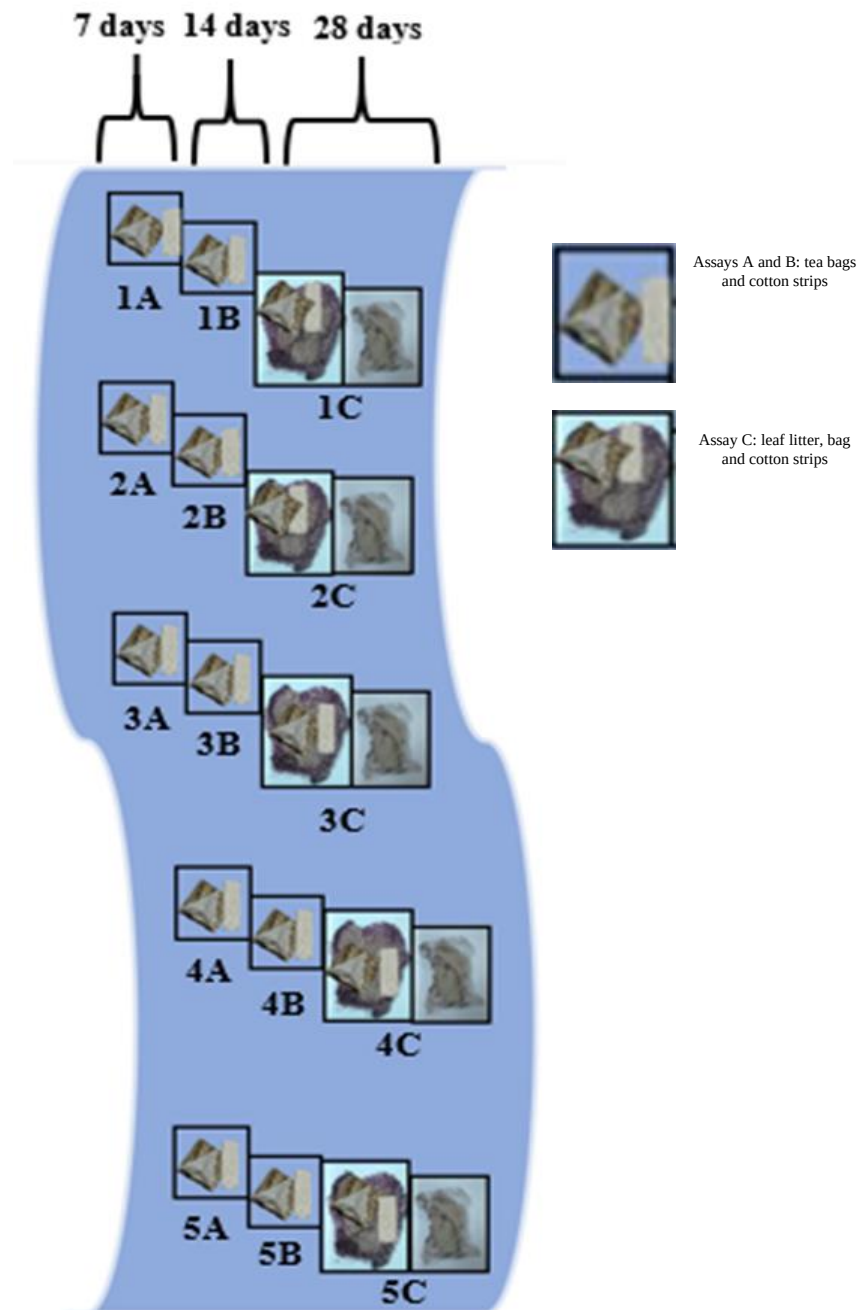


Figure 4.5. Diagram displaying the deployment order for each reach, where 1A-5A show coarse mesh bags containing tea bags (~3 g) and cotton strips (~0.8 g) deployed for 7 days; 1B-5B display coarse mesh bags containing tea bags (~3 g) and cotton strips (~0.8 g) deployed for 14 days; and 1C-5C display coarse mesh bags containing tea bags (~3 g), cotton strips (~0.8 g), coarse bags containing leaf litter (5 ± 0.01 g) and fine bags containing leaf litter (3 ± 0.01 g).

4.2.3 Laboratory Analysis

4.2.3.1 Physicochemical Analysis

Water samples were passed through 0.22 µm membrane filters (Fisher) and transferred into 50 ml falcon tubes for analysis of total organic carbon (TOC) and nitrate (NO_3^-). Samples were stored in the fridge prior to analysis. In the laboratory, 20 ml of each sample was transferred into glass vials and run on the TOC autosampler (TOC-L ASI-L Shimadzu). Since total carbon (TC) is the combination of both total organic carbon (TOC) and total inorganic carbon (TIC), non-purgeable organic carbon (NPOC) represented TOC following TIC removal (Onogbosele, 2015; Shimadzu total organic carbon analyser ASI-L). NO_3^- concentrations were measured using ion chromatography.

4.2.3.2 Leaf Litter

Leaves were air dried for at least 14 days and weighed to the nearest 0.0001 g. To calculate mass loss, a sub-sample of each leaf was weighed, placed into a crucible and combusted at 550°C for over 4 hours to determine loss on ignition (LOI). The percentage of mineral content remaining in the crucible was subtracted from the post-field leaf loss to determine relative mass loss (minus mineral aggregates).

4.2.3.3 Tea Bags

Tea bags were air dried and weighed. The contents of each bag were emptied, weighed and ashed at 550°C for at least 4 hours to determine LOI. The percentage of mineral content remaining in the crucible was subtracted from the post-field leaf loss to determine relative mass loss (minus mineral aggregates).

4.2.3.4 Cotton Strips

Cotton strips were air dried and weighed to the nearest 0.0001 g. Strips were examined in an external laboratory; the end 1 cm of each strip was placed on a tensiometer (Mark-10 brand, Model #MG100, Copiague, NY, USA) and pulled apart at a rate of 2 cm/min

to determine maximum tensile-strength (lbs) (Tiegs *et al.*, 2013). Careful attention was made not to make the strips too tight or too loose to avoid ripping and slipping.

4.2.4 Data Analysis

4.2.4.1 Physicochemical Analysis

Physicochemical parameters were compared between restored and unrestored sites across rivers and reaches using a two-way ANOVA on ranks, due to the lack of normality and non-equal variance in the dataset, followed by a Tukey's post-hoc test. All statistics were performed using SigmaPlot 14.0.

4.2.4.2 Leaf Litter and Tea Bags

Leaf litter breakdown is measured as mass loss over time and is measured by fitting an exponential decay model (Benfield, 2006). Decomposition rates were calculated using the following equation:

$$\text{Rate of decay (kdd)} = \frac{\ln \left(\frac{\text{Final mass}}{\text{Initial mass}} \right)}{\text{Degree day}}$$

Equation 4.1.

Where the rate of decay (k_{dd}) is determined by the final corrected mass (g) of leaves and tea bags, divided by the initial mass and scaled by breakdown rates (estimated degree day).

As with physicochemical data, leaf litter, tea bag and strip data was analysed by comparing restored and unrestored sites across rivers and reaches using a two-way ANOVA on ranks, due to the lack of normality and non-equal variance in the dataset, followed by a Tukey's post-hoc test.

4.2.4.3 Cotton Strips

Tensile-strength loss was shown as percent lost per degree day to rule out the effects of temperature among sites and rivers:

$$\% \text{ Tensile Strength Loss (dd)} = \frac{\left(\frac{\text{Tensile Strength Treatment Strips}}{\text{Tensile Strength Reference Strips}} \right)}{\text{Degree day}}$$

Equation 4.2.

Temperature corrected tensile-strength loss values, where percentage loss is the maximum tensile-strength recorded for each of the strips incubated in the field and tensile-strength reference strips are the mean strength of 10 strips that were not incubated in the field but cleaned with ethanol and stored in a desiccator (control tensile strength 67.3 lbs $\pm$ 0.91, n = 10).

4.3 Results

4.3.1 Physical Habitat

Velocity varied significantly between rivers ($F_{4,589} = 12.15$; $p < 0.001$) and reaches ($F_{1,589} = 18.01$; $p < 0.001$). Except for the Brent, all other reaches exhibited significantly higher velocities at restored reaches (Table 4.1). The Hogsmill and Wandle demonstrated the highest degree of difference between reaches ($p < 0.001$), where values ranged from 0.23 m s⁻¹ (Hogsmill) to 0.35 m s⁻¹ (Wandle). Significant differences were also observed in terms of interactions between rivers and reaches ($F_{4,589} = 18.14$; $p < 0.001$). Mean depths differed significantly among rivers ($F_{4,588} = 50.51$, $p < 0.001$), ranging from 7.35 cm (Hogsmill) to 19.57 cm. The Brent was the only river to show equivocal differences in depth amongst paired reaches. pH values were significantly different across rivers ($F_{4,190} = 9.80$; $p < 0.001$), ranging from 7.71 to 8.08 (Table 4.1). Values did not differ between reaches ($F_{1,190} = 1.83$; $p = 0.18$). Similarly, dissolved oxygen (%) differed between rivers ($F_{4,190} = 15.67$; $p < 0.001$). The lowest and highest values were observed at the Brent restored reach (101.38%) and Hogsmill restored reach (118.45%), respectively. Differences were not observed between reaches ($F_{1,190} = 2.48$; $p < = 0.12$), and with interactions between rivers and reaches ($F_{4,190} = 1.35$; $p = 0.26$). Conductivity differed

significantly between rivers ($F_{4,190} = 15.59$; $p < 0.001$), ranging from 774.80-775.40 (Ravensbourne) to 1080.05-1082.30 (Brent). No differences were observed between reaches ($F_{1,190} = 0.01$; $p = 0.92$) or river-reach interactions ($F_{4,190} = 0.03$; $p = 0.998$). Temperatures differed between rivers ($F_{4,189} = 17.35$, $p < 0.001$). Values were lowest at the Hogsmill (5.62°C) and highest at the Wandle (7.61°C). Temperatures did not vary between reaches ($F_{1,189} = 0.54$; $p = 0.47$) or interact with rivers ($F_{4,189} = 0.25$, $p = 0.91$). TOC differed between rivers ($F_{4,173} = 8.29$; $p < 0.001$), with the lowest and highest concentrations at the Wandle (1.70-1.82 mg l⁻¹) and Brent (4.51-4.79 mg l⁻¹), respectively. Concentrations did not differ between reaches ($F_{1,173} = 1.52$, $p = 0.22$) or river-reach interactions ($F_{4,173} = 1.01$; $p = 0.40$). The Ravensbourne was the only river to display reach differences (4.27 mg l⁻¹). Nitrate concentrations ranged from 9.46 mg l⁻¹ to 13.93 mg l⁻¹ but did not differ between rivers ($F_{4,177} = 2.01$; $p = 0.07$), restoration ($F_{1,177} = 0.10$; $p = 0.75$), or river-reach interaction ($F_{4,177} = 0.35$; $p = 0.85$).

Table 4.1. A summary of physicochemical characteristics across sites for velocity (m s^{-1}), depth (cm) ($n = 60$), pH, dissolved oxygen (%), conductivity ($\mu\text{s cm}^{-1}$), temperature, total organic carbon (TOC) and nitrate (NO_3^-) ($n = 20$). Values in parentheses denote one standard error. Significant differences between restored and unrestored reaches are in bold; differences among rivers are indicated by letter groupings.

Parameter	Ravensbourne		Pool		Wandle		Hogsmill		Brent	
	Rest.	Unrest.	Rest.	Unrest.	Rest.	Unrest.	Rest.	Unrest.	Rest.	Unrest.
Velocity (m s^{-1})	0.29^d	0.20^d	0.21^{b,a}	0.09^{b,a}	0.35^a	0.18^a	0.23^{c,a}	0.12^{c,a}	0.12^a	0.33^a
	(0.03)	(0.03)	(0.01)	(0.01)	(0.03)	(0.01)	(0.02)	(0.02)	(0.03)	(0.04)
Depth (cm)	9.35^a	20.54^a	16.55^e	31.12^e	19.55^{c,b}	16.69^{c,b}	7.35^b	14.38^b	19.57 ^{d,c}	17.01 ^{d,c}
	(0.56)	(0.78)	(0.91)	(1.24)	(0.66)	(0.62)	(0.36)	(0.81)	(1.51)	(1.37)
pH	8.08 ^{a,d}	8.05 ^{a,d}	7.91 ^c	7.76 ^c	7.80 ^{b,d}	7.71 ^{b,d}	8.05 ^d	7.96 ^d	7.80 ^d	7.90 ^d
	(0.03)	(0.04)	(0.02)	(0.14)	(0.02)	(0.03)	(0.05)	(0.04)	(0.04)	(0.06)
DO (%)	111.06 ^d	107.94 ^d	106.29 ^{b,d}	103.16 ^{b,d}	107.53 ^{c,b}	103.78 ^{c,b}	118.45 ^a	115.06 ^a	101.38 ^b	105.12 ^b
	(0.98)	(1.41)	(0.79)	(0.67)	(2.31)	(1.51)	(3.30)	(3.16)	(1.51)	(1.66)
Cond. ($\mu\text{s cm}^{-1}$)	774.80 ^a	775.40 ^a	838.80 ^d	833.05 ^d	802.30 ^c	801.60 ^c	852.97 ^a	874.25 ^a	1082.30 ^b	1080.05 ^b
	(16.55)	(17.11)	(13.57)	(15.56)	(56.81)	(57.24)	(41.51)	(13.38)	(66.47)	(72.43)
Temp ($^{\circ}\text{C}$)	6.66 ^d	6.67 ^d	6.20 ^d	6.03 ^d	7.48 ^a	7.61 ^a	5.87 ^{b,d}	5.62 ^{b,d}	6.06 ^c	5.78 ^c
	(0.25)	(0.25)	(0.23)	(0.20)	(0.09)	(0.09)	(0.39)	(0.43)	(0.11)	(0.10)
TOC (mg l^{-1})	4.27^{a,c,d}	2.69^{a,c,d}	2.85 ^{d,b,c}	2.23 ^{d,b,c}	1.70 ^{b,c}	1.82 ^{b,c}	2.30 ^c	2.59 ^c	4.79 ^a	4.51 ^a
	(0.95)	(0.39)	(0.41)	(0.46)	(0.19)	(0.22)	(0.28)	(0.46)	(0.76)	(0.76)
NO_3^- (mg l^{-1})	11.75	11.94	10.10	11.38	12.65	13.93	13.03	12.77	10.73	9.46
	(1.20)	(1.10)	(1.21)	(1.17)	(1.33)	(1.35)	(1.62)	(1.79)	(1.25)	(1.17)

4.3.2 Coarse Mesh Bags

Breakdown rates ranged from ~ 0.0007 to $0.001\ k_{dd}$ (Table 4.2, rounded to 3 decimal places), however no overall statistical differences between study rivers ($F_{4,27} = 2.17$; $p = 0.10$), paired reaches ($F_{1,27} = 2.82$; $p = 0.10$), or river/reach interactions ($F_{4,27} = 0.95$; $p = 0.45$) were found. Decomposition progressed very slowly in coarse mesh bags across all rivers and reaches, with the Ravensbourne, Pool, Hogsmill and Brent failing to reach 20% mass loss after 28 days (Figure 4.6). The highest and lowest mass loss were recorded at the restored Wandle (22%) and Hogsmill reach (9.28%), respectively (Table 4.2; Figure 4.6). Handling losses were 2.98% for coarse mesh bags.

4.3.3 Fine Mesh Bags

Decay rates ranged from 0.0001 (Ravensbourne unrestored) to $0.001\ k_{dd}$ (Brent / Ravensbourne restored) (Table 4.2). Two-way ANOVA results revealed that rates did not vary between rivers ($F_{4,22} = 1.97$; $p = 0.14$). However, reach-scale analyses revealed significant differences between paired reaches ($F_{1,22} = 7.11$; $p = 0.01$), due to elevated rates of decay at the Brent restored reach ($p = 0.009$). Mass loss was significantly lower in fine mesh bags. For example, after 28 days, mass loss did not exceed 10% across all reaches except for the Brent restored reach (82.76% remaining). The lowest percentage of mass loss was observed at the Ravensbourne unrestored reach (97.94% remaining) (Figure 4.7). Percentage loss differed between reaches, with higher mass loss at the Brent restored reach. Handling losses were 2.45% for fine mesh bags.

4.3.4 Tea bag Decomposition

Overall, rates of decomposition among tea bags were faster than that of leaf litter, ranging from 0.001 to 0.002 k_{dd} (Table 4.2). Rates of loss did not vary among rivers ($F_{4,121} = 0.71$; $p = 0.59$) or between reaches ($F_{1,121} = 0.18$; $p = 0.67$). No significant interactions were observed between rivers and reaches ($F_{4,121} = 1.09$; $p = 0.37$). Rates of mass loss were slow among tea bags, with the Pool, Hogsmill and Brent failing to exceed 20% mass loss after 28 days (Table 4.2; Figure 4.8). The restored Ravensbourne site and both sites at the Wandle exceeded mass loss beyond 20% after 28 days, with values falling to 75%, 79% and 76%, respectively. Percent mass remaining after 28 days revealed no marked differences between rivers (Figure 4.8). The average breakage bag loss for teabags was 2.85%.

4.3.5 Cotton Strip Tensile-Strength Loss

Temperature corrected tensile-strength loss values ranged from 0.23 to 0.36% loss/degree day (Table 4.2). Rates did not vary between rivers ($F_{4,133} = 0.20$; $p = 0.94$), or reaches ($F_{1,133} = 0.04$; $p = 0.83$), or among river-reach interactions ($F_{4,133} = 0.62$; $p = 0.65$). Percentage tensile-strength of cotton strips remaining after 28 days revealed markedly higher breakdown than that of leaf litter and tea bags, with the lowest percentage remaining observed at the Wandle restored (5.42%) and Brent restored reaches (32.32%) (Table 4.2; Figure 4.9). Tensile-strength remaining after 28 days ranged from 5.41 to 34.68 lbs. At the paired Wandle reach, loss was notably higher at the restored reach. Tensile-strength remaining was also lower at the Brent unrestored reach, where tensile-strength values lower than 3 lbs were recorded.

Table 4.2. A summary of breakdown rates (k_{dd}) for coarse, fine mesh bags and tea bags (mass loss) and cotton strips (% tensile-strength loss (k_{dd})). Values in parentheses denote one standard error. Significant differences between restored and unrestored reaches are in bold; differences among rivers are indicated by letter groupings.

Sample	Ravensbourne		Pool		Wandle		Hogsmill		Brent	
	Rest.	Unrest.	Rest.	Unrest.	Rest.	Unrest.	Rest.	Unrest.	Rest.	Unrest.
Coarse Bags	0.001 (0.0001)	0.001 (0.0002)	0.001 (0.0001)	0.001 (0.0001)	0.001 (0.0002)	0.001 (0.001)	0.001 (0.00004)	0.001 (0.0001)	0.001 (0.0002)	0.001 (0.0002)
Fine Bags	0.001 ^{a,b} (0.0003)	0.0001 ^{a,b} (0.00002)	0.001 ^{a,b} (0.0004)	0.001 ^{a,b} (0.0001)	0.0003 ^a (0.0001)	0.0003 ^a (0.0001)	0.0004 ^{a,b} (0.0002)	0.001 ^{a,b} (0.0001)	0.001^b (0.0002)	0.0003^b (0.00003)
Tea Bags	0.002 (0.0002)	0.001 (0.0002)	0.002 (0.0001)	0.002 (0.0002)	0.002 (0.0002)	0.002 (0.0002)	0.002 (0.0004)	0.001 (0.0002)	0.002 (0.0002)	0.002 (0.0003)
Cotton Strips	0.25 (0.06)	0.26 (0.05)	0.29 (0.06)	0.28 (0.06)	0.31 (0.06)	0.23 (0.06)	0.25 (0.05)	0.25 (0.07)	0.24 (0.08)	0.36 (0.09)

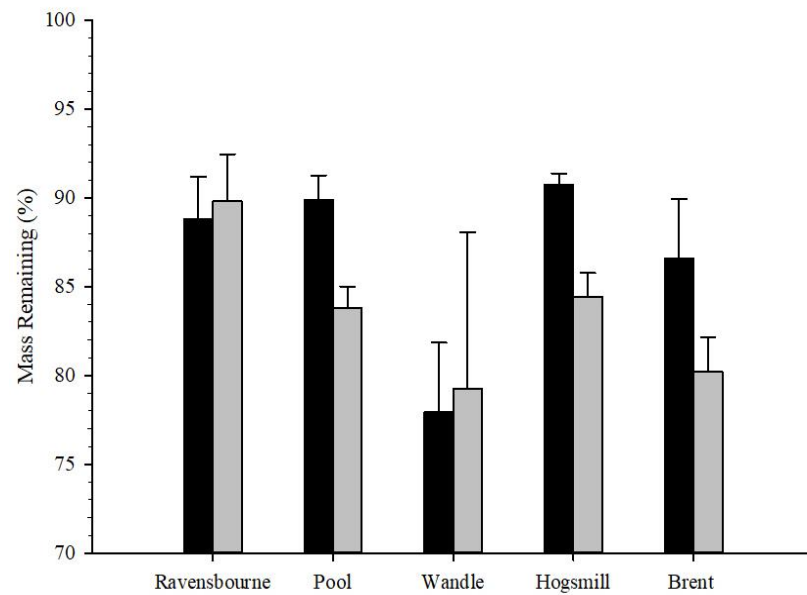


Figure 4.6. Percentage mass remaining for leaf litter in coarse mesh bags after 28 days. Black columns denote restored sites; grey columns denote unrestored sites.

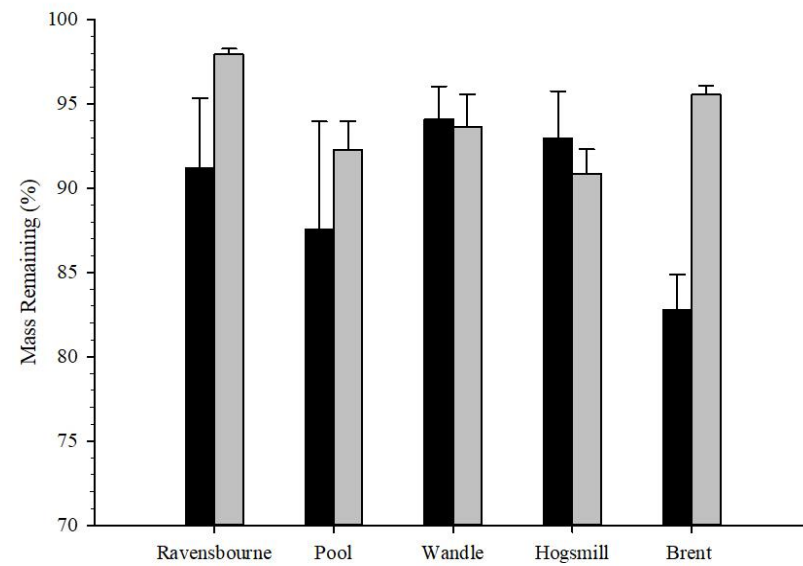


Figure 4.7. Percentage mass remaining for leaf litter in fine mesh bags after 28 days. Black columns denote restored sites; grey columns denote unrestored sites.

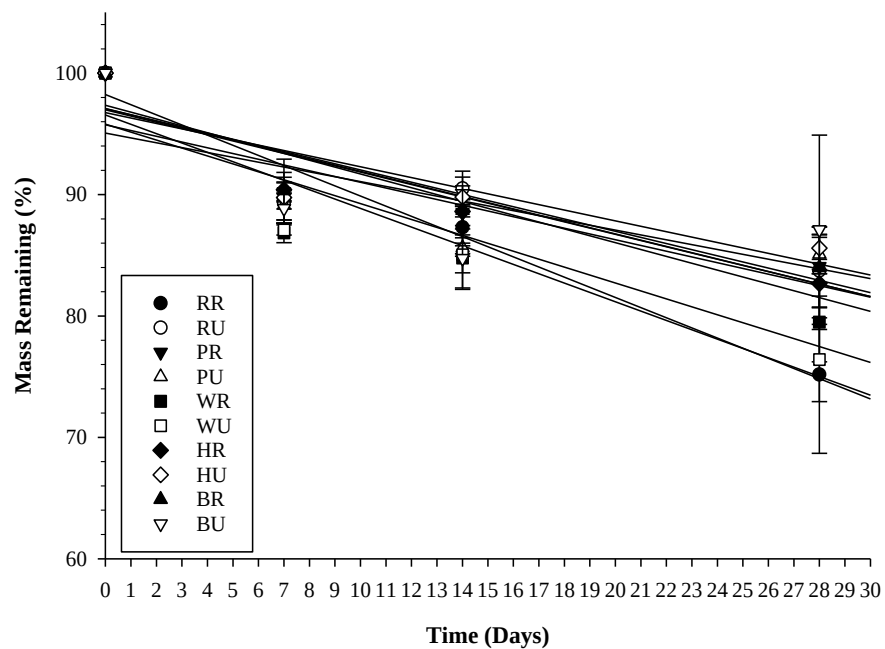


Figure 4.8. Percentage mass remaining for tea bags after 7, 14 and 28 days.

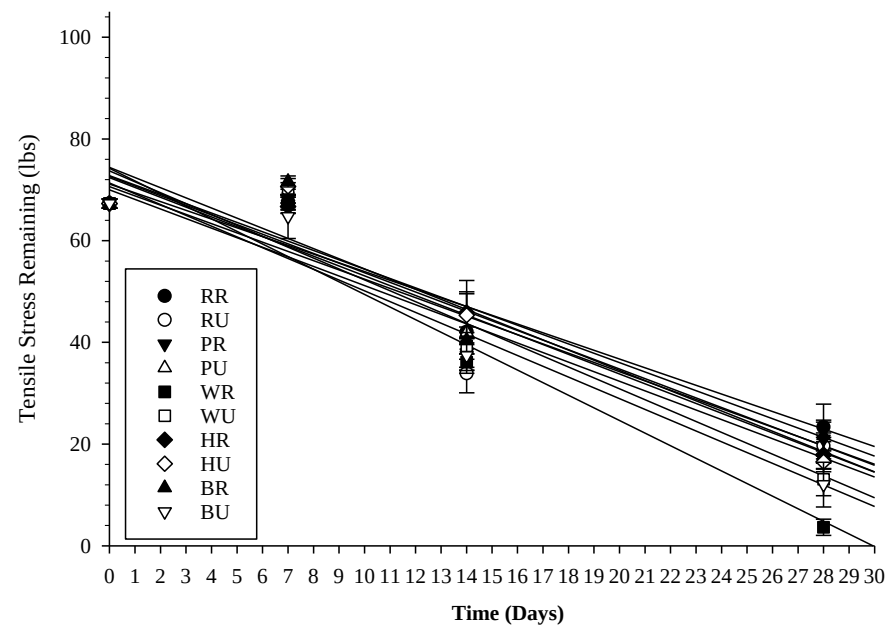


Figure 4.9. Tensile-strength (lbs) remaining for cotton strips after 7, 14 and 28 days.

4.4 Discussion

The overall aim of this study was to compare decomposition rates across restored and unrestored reaches using multiple assay methods. The objectives were to investigate differences between reaches and to evaluate the effectiveness of using different assays to measure decomposition. It was hypothesised that decomposition would differ significantly between restored and unrestored sites, owing to processes including physical abrasion, nutrient processing and biological activity. Furthermore, it was hypothesised that cellulose cotton strips would provide a preferential tool to measure decomposition over leaf litter and tea bags due to their uniformity and ease of application.

Results revealed an overall consistent linear response in decomposition rates across rivers and reaches (restored and unrestored), thus rejecting the first hypothesis outlined in this study. Leaf litter breakdown results from this study are consistent with other temperature studies of London streams (0.001-0.012 k_{dd}) which reveal equivocal responses to restoration (*sensu* Smith and Chadwick, 2014). Given the urban nature of these streams where nutrient loading, impervious surface cover (reducing litter supply) and hydrological regimes (accelerating litter transport) are likely to override restoration efforts (Walsh *et al.*, 2005), it is perhaps not surprising that no consistent differences were observed in decomposition rates between rivers and reaches. However, previous studies have identified inverse relationships between leaf litter breakdown and impervious surface cover (Chadwick *et al.*, 2006; Young and Collier, 2009). There is limited research to show that promoting channel heterogeneity can render biological recovery, however geomorphological restoration practices continue to be a highly popular restoration strategy (Palmer *et al.*, 2010). As previous studies have noted, focusing on reach-scale improvements seems logical given the many logistical and administrative restrictions on urban rivers (Smith and Chadwick, 2014), nevertheless, this will not necessarily accrue ecological benefits. Changes in habitat such as substrate type or percentage cover can

affect reach-scale decomposition (Fenoy *et al.*, 2016). Macrophytes trap sediment containing organic matter (dissolved organic carbon and fine particulate organic carbon) which provides a food source for shredders, in turn increasing decomposition.

For coarse leaf litter bags, no differences were observed between rivers or reaches (restored and unrestored). Observed litter decomposition rates were within the range of previous work but also lower than that of previously reported values of Smith and Chadwick (2014) which ranged from 0.001 to 0.01 k_{dd} (Table 4.2). This may be attributed to reduced temperatures inhibiting microbial and invertebrate activity. Mass loss remaining in this study did not exceed 25% after 28 days, indicative of reduced microbial activity. Higher rates of decomposition among coarse bags compared with fine bags and tea bags could suggest that breakdown might be attributed to invertebrate shredding. Previous studies have noted increased decomposition rates alongside elevated densities of macroinvertebrates such as oligochaetes which feed on FPOM (Pascoal and Cassio, 2004). Although velocity was significantly higher at restored sites, it was not considered to have significantly influenced breakdown among litter bags, as seen in previous studies (Chadwick *et al.*, 2006). A possible exception could be the Wandle, where marginal increases in coarse leaf % mass remaining (77%) may have been attributed to physical abrasion from increased velocities.

Breakdown rates in fine mesh bags were lower than that of other materials, ranging from 0.0001 to 0.001 k_{dd} . This is possibly a result of invertebrate isolation and protection from physical fragmentation which is more likely in coarse mesh bags. The Brent was the only river to exhibit reach differences among fine bags, with higher levels of breakdown at the restored reach (Table 4.2). Previous studies have detected the presence of oil contamination along this stretch of river, a presumed inhibitor of breakdown (Smith and Chadwick, 2014). Previous studies within this project have recorded high levels of

ammonium, which have the potential to reduce or accelerate breakdown. Microbial activity is an important factor influencing breakdown in temperate streams, however the role of bacteria and fungi in decomposition necessitate further research.

Tea bag decomposition rates were most similar to coarse leaf bags (0.001 to 0.002 k_{dd} .) and comparable to recorded values by Smith and Chadwick (2014). Higher rates of decomposition among tea bags relative to fine leaf litter bags are noteworthy given their similar physical composition (both excluding macrofauna). Rooibos tea bags are known to decompose at faster rates than other organic matter materials (i.e. green tea) (Djukic *et al.*, 2018). Together these suggest that tea bags may provide a preferential food source for microbes and a more effective assay for monitoring decomposition in aquatic systems.

Decay occurred more readily among cotton strips, reflecting their sensitivity to environmental conditions and preference as a carbon source (Tiegs *et al.*, 2013). These findings support the second hypothesis of this study which postulated that cotton strips would provide a preferential tool to measure decomposition over leaf litter and tea bags. Previous studies of tensile-strength corrected losses (% loss/degree day) have reported lower values than those reported in this study, with losses ranging from 0.11% dd and 0.13% dd and a mean of 0.12% dd (Tiegs *et al.*, 2013). Given that significant differences between reaches were only revealed for cotton strips after 28 days (Wandle and Brent), it is likely that the incubation times in this study were not long enough to highlight differences among leaf litter and tea bags. After 28 days, cotton strip decomposition was highest at the Wandle (Figure 4.9; 4.10) and Brent, a possible result of elevated water temperatures and pollution, respectively. Together these highlight the need to further understand the compensatory effects occurring, which influence the survival of invertebrates and microbes essential for breakdown (Pascoal and Cassio, 2004). Nutrients can stimulate microbial activity in moderate concentrations, however heavy metal

pollution can also reduce invertebrate feeding (Woodward *et al.*, 2012). Previous studies have attributed changes in breakdown among strips to ammonium ($p = 0.023$) (Boulton and Quin, 2000; Tiegs *et al.*, 2013) and substrate ($p = 0.021$), where values were higher in streams dominated by gravel (Tiegs *et al.*, 2013).



Figure 4.10. River Wandle cotton strip samples after 7 days and 28 days.

Physical habitat characteristics were generally consistent with ecological restoration aims characteristic of increased velocities and reduced depths (Table 4.1). However, these did not yield improvements in ecological function, concurring with the posited argument put forward by Palmer *et al.* (2010) that geomorphological-based restoration is largely detached from ecological theory. In addition, the results differed from other studies undertaken at these sites as part of this research project, where overall flows were more homogenous between reaches, highlighting the importance of repeat studies.

Physicochemical parameters, including pH, dissolved oxygen, conductivity and temperature did not differ between restored and unrestored sites. (Table 4.1). However notable site differences were observed, including increased conductivity levels at the Brent (1080.05-1082.30 $\mu\text{S cm}^{-1}$) and higher temperatures at the Wandle (7.48-7.61°C), both of which differed to other sites (but not between paired reaches). It is possible that increased temperatures at the Wandle may have stimulated breakdown. For TOC, the Ravensbourne was the only river to show a significant difference between reaches (2.69 mg l^{-1} to 4.27 mg l^{-1}), however this had no observed influence on breakdown. TOC differed among sites, exhibiting the highest concentrations at the Brent. NO_3^- concentrations did not differ between rivers or reaches, reducing the likelihood of additional nitrogen stimulating growth (Irons *et al.*, 1988).

4.5 Conclusions

Altogether, results from this study highlighted variability among experimental materials, sites and on occasion, reaches. However, overall leaf litter decomposition revealed equivocal responses to restoration between restored and unrestored sites. It is difficult to determine the drivers of change, and previous studies have attributed variable stream responses to possible stochastic and unpredictable events (Smith and Chadwick, 2014), also defined as “impingement of chance events on an experiment in progress” (Hurlbert, 1984). In urban London rivers, pollution events, misconnections and flashy events are likely to override restoration efforts that have been put in place. This is the case for the Brent where high ammonium concentrations were previously detected along this stretch of river (Chapter 3). The findings from this study necessitate ongoing work to decipher the contribution of hydrology, water quality and invertebrates / microbes to decomposition.

Chapter 5: Fish and Macroinvertebrate Community Responses to Restoration in Urban Tributaries of the River Thames

Abstract

Geomorphological modifications, hydrological change and nutrient pollution are among the dominant insults that jeopardise ecological integrity in urban streams. Some restoration projects carry the assumption that manipulating channel geomorphology to mimic pre-degraded conditions can foster ecological recovery – the “field of dreams” hypothesis. In this study, fish and macroinvertebrate community structure were examined across five paired restored and unrestored tributaries of the River Thames using electric fishing surveys, surber sampling and broad physical habitat assessments. Fish abundance was higher at restored sites (955 individuals) compared with unrestored sites (586 individuals), largely driven by the presence of European bullhead *Cottus gobio* at the Wandle and Hogsmill. Restored sites exhibited higher overall fish biomass (restored mean: 13.40 g, unrestored mean: 5.89 g), attributed to the presence of Brown trout *S. trutta* at the Wandle. Macroinvertebrate biomass, taxa richness and Shannon-Weiner index values differed significantly between rivers and reaches (restored and unrestored). Results revealed that these differences were largely driven by elevated values at the Wandle and Hogsmill restored sites. Non-metric Multi-Dimensional scaling (NMDS) results for invertebrate taxa abundance revealed a marginally significant stress value between sites (stress = 0.18). These findings highlight the merit of reach-scale restoration projects for supporting higher abundances of sensitive fish species (e.g. *S. trutta*) and invertebrates. However, given the scope of reach-scale projects in London, it seems clear that ecological recovery is restricted by catchment-scale processes and stochastic events. Future studies should investigate the importance of water quality, aquatic vegetation and habitat-scale hydromorphology as factors influencing restoration success.

5.1 Introduction

5.1.1 Fish

Fish and macroinvertebrates are useful measures of stream health as they are easy to identify and respond to in-stream water quality and hydrology (Roni *et al.*, 2008; Lorenz *et al.*, 2013). Fish are mobile, long-lived and require several habitat types during their life cycle (Crisp, 1996). In early life, specific substrata are critical for supporting spawning grounds for fish guilds, while fish larvae and juveniles depend on nursery habitats (Cowx and Welcomme, 1998). Species including European eel *Anguilla anguilla* and brown trout *S. trutta* utilise riffles, pools and channel banks for upstream migration and shelter. Complex and diverse habitats are critical for improving resilience to physical barriers and chemical pressures (e.g. weirs and pollution) (Lorenz *et al.*, 2013).

Hydromorphological processes are critical for supporting fish and invertebrate communities (Pretty *et al.*, 2003). Hydrological alterations, sediment alterations and nutrient pollution can degrade fish habitats by modifying leaf litter composition, structure of coarse woody debris and sediment dynamics (Niezgoda and Johnson, 2005; Miller and Kochel, 2010; Čivas *et al.*, 2016; Teichert *et al.*, 2016). Episodic flood events are associated with the scouring away of mineral substrata and displacement of macrophytes which reduces physical habitat availability for fish species. They also alter invertebrate habitats which are important food sources for insectivorous fish species (Stewardson *et al.*, 2016).

Growth rates and body size among fish species are influenced by several biotic and abiotic factors including temperature, forage availability and intraspecific species competition (Alanara *et al.*, 2001; O’Gorman *et al.*, 2016). Some studies have highlighted significant differences in growth rates in response to seasonality, temperature and illuminations which can lead to the developments of annuli and circuli on fish scales (Brown, 1951).

Thermal stress has been found to affect larval development for *A. anguilla*, limiting development below 16°C, accelerating growth development between 16 and 22°C and leading to mortality at temperatures over 24°C (Politis *et al.*, 2017). Other pressures, including the ubiquitous nature of fine sediment, can influence fish health through physical organ damage (Kemp *et al.*, 2011). Furthermore, fine sediment can jeopardise developmental stages of lithophilic fish sensitive to sand and clay particles in spawning grounds, limiting reproduction success for lithophilic fish. Finally, sediment can negatively impact flows, reducing oxygen permeation, metabolic waste removal and fry emergence (Pattison *et al.*, 2015; Sear *et al.*, 2016).

5.1.2 Macroinvertebrates

Macroinvertebrates are pervasive in riverine environments, easy to collect and responsive to pollution (Huddart *et al.*, 2016). They are often a preferred indicator of stream health due to their sedentary nature, allowing for prolonged exposure to chemical conditions present in sediment and water (Miller and Kochel, 2010). Invertebrates provide a critical food resource for littoral and pelagic fish and birds, therefore exposure to poor sediment and water quality can influence species development and functional feeding patterns (Woodward and Warren, 2007). In urban streams, invertebrates are often affected by inputs of inorganic nitrogen (N) which can reduce biotic diversity and increase the dominance of tolerant species (Grimm *et al.*, 2005; Wollheim *et al.*, 2005). The lifespan of invertebrates, although shorter than that of fish, can reflect environmental conditions and highlight a graded response to acting pressures (Visser *et al.*, 2017).

5.1.3 Restoring Ecological Integrity

Many river restoration projects aim to restore hydrological heterogeneity, regulate sediment distribution and reduce nutrient loads (Bernhardt and Palmer, 2007; Stranko *et al.*, 2008; Hines and Hershey, 2011). These often take the form of re-shaping rivers

through natural hydraulic energy or altering channel geomorphology (Hohensinner *et al.*, 2018). Common approaches involve engineering gravel riffles and boulders to promote oxygenation for habitat recovery (Pretty *et al.*, 2003; Bukaveckas, 2007). During the 1920s, restoration structures were commonplace in high-gradient gravel salmonid streams of the USA (Brookes, 1996). These have been increasingly adopted in UK rivers to encourage habitat recovery for fish and invertebrates (Brookes *et al.*, 1996). Many restoration techniques adopted in the UK have been created with the intention of fostering functional ecological environments (Pretty *et al.*, 2003; Harrison *et al.*, 2004). Palmer *et al.* (1997) postulated that manipulating channel geomorphology to mimic pre-degraded conditions could foster re-colonisation of aquatic organisms and encourage ecological integrity – the “field of dreams” hypothesis. The tendency to address WFD classification elements by complying with certain standards which assume ecological recovery (Giakoumis and Vouvoulis, 2019), is a likely contributor towards WFD failures.

Increasingly, restoration has been geared towards reducing fine sediment through approaches such as gravel augmentation and implementing in-stream structures (e.g. woody debris, boulders) (Osei *et al.*, 2015). Gravel augmentation involves replenishing depleted or degraded sediment to increase spawning substrate availability (Pander *et al.*, 2015). Hydraulics can be manipulated through implementation of in-stream structures such as boulders and wood to encourage hyporheic water exchange. Furthermore, in-stream gravel cleansing techniques such as substrate raking trigger mechanical bioturbation of sediments, promoting fine sediment mobilisation and improving spawning gravels for lithophilic fish (Pander *et al.*, 2015). Jetting is an increasingly popular method used to cleanse gravels of fine sediment and increase habitat preferences for species. In a study examining the impacts of jetting on sediment composition in the Great Ouse (UK), gravels were removed at the riffle-scale, resulting in coarser and better sorted sediment,

but effects were short-lived (<12 months). Despite providing local benefits, jetting has been increasingly criticised for leading to sedimentation downstream and causing over-scouring (Sternecker *et al.*, 2013).

The overarching aim of this study was to deliver a quantifiable assessment of fish and invertebrate community structure as an indicator of overall stream health. The main objectives were **(1)** To assess fish abundance, biomass and richness across restored and unrestored reaches using 3-sweep catch depletions, **(2)** to examine macroinvertebrate community structure across restored and unrestored reaches using surber sampling techniques, and **(3)** to compare broad physical habitat characteristics between restored and unrestored sites. It was hypothesised that **(1)** fish abundance, biomass and richness would increase at restored sites due to improvements in hydromorphological conditions fostering ecological integrity **(2)** macroinvertebrate density, biomass and richness would increase at restored sites due to improvements in water and habitat quality, and **(3)** physical habitat heterogeneity would increase at restored sites due to improvements in hydrological heterogeneity and habitat diversity.

5.2 Materials and Methods

Hydromorphological assessments, electric fishing surveys and surber samples were carried out across five paired restored and unrestored tributaries of the River Thames. Paired restored and unrestored reaches were selected within a 50 - 200 m range of one another, close enough that water quality is unlikely to have varied significantly, yet far enough apart for differences in hydrology and in-stream vegetation to be assessed. Despite variations in restoration approaches, a common aim was to promote hydrological connectivity and flow heterogeneity to augment habitat diversity.

5.2.1 Field Sampling

5.2.1.1 Physical Habitat

Five transects were placed at equal distances along each 25 m reach. Five repeat measurements were obtained for width (every 5 m), depth and velocity (every 1 m) across each width transect using a standard tap method.

5.2.1.2 Electric Fishing Three-Sweep Depletions

A 25 m sampling section was selected for assessment along restored and unrestored reaches to incorporate the restoration activity and account for differences in restoration length. Electric fishing surveys were conducted using a LR-20B, which was set at ~400-1000 secs; 40 Hz, 150-200 v. Settings were adjusted at each site for optimal catch efficiency due to high conductivity (ranging between 591 to 730 $\mu\text{S cm}^{-1}$). Fish welfare remained paramount throughout sampling. Stop nets were deployed at either end of the reach to create a closed population and prevent fish migration or emigration and a three-sweep catch-depletion method was used to estimate fish populations. All fish were identified to species level, measured for length (fork length mm) and weight (g) and replenished below the downstream limit following each sweep. Where possible, scales were obtained for ageing (*S. trutta*, *Leuciscus leuciscus* and European Perch *Perca fluviatilis*). Surveys were carried out in the early autumn (September – October 2016) to allow for the examination of young-of-the-year (YOY) fish.

5.2.1.3 Macroinvertebrate Community Structure: Surber Sampling

Five surber samples (0.33 m x 0.33 m; mesh size 250 μm) were obtained randomly (determined by stream access). All invertebrates were obtained by disturbing the top 5 cm of sediment and transferred into glass bottles containing industrial methylated spirit (90%) for preservation.

5.2.2 Laboratory Analysis

In the laboratory, invertebrate specimens were placed onto a Petri dish for examination. Using a high specification microscope, taxa were sorted by species, except for *Oligochaeta* spp., *Chironomidae* spp, *Limnephilidae* spp. and *Polycelis* spp., which were classified to family level.

5.2.3 Data Analysis

5.2.3.1 Physical Habitat

Average wetted width, depth and velocity were compared between restored and unrestored reaches across each river and between rivers, using a two-way ANOVA on ranks, due to the lack of normality and non-equal variance in the dataset, followed by a Tukey's post-hoc test. Standard error was recorded to scale standard deviation based on sample size.

5.2.3.2 Fish Population Estimates and Biomass

Estimated fish populations were calculated using the Carle and Strub (1978) catch depletion method. This method was chosen to account for data of species that did not conform to the maximum likelihood method (Carle and Strub, 1978). Density, biomass and richness are indicative of community structure (Saint-Germain *et al.*, 2007; Smith and Chadwick, 2014) and can reflect a response to hydromorphological and chemical conditions. For biomass, unknown fish weights (< 1 g) were determined using length-weight coefficients based on data of collective species caught during the survey. Biomass calculations were used to attribute proportional masses of certain species to overall taxa (Howard and Cuffey, 2006). The Fisheries Stock Assessment (FSA) package (Ogle, 2016) was installed in R Studio and coded to denote a string of data corresponding to each sweep. The output revealed the catch efficiency (*p* value) and 95% confidence values. For species where estimated biomass was not recorded (i.e. *S. trutta*) Fulton's

conditions factor (k) was used to determine fish health based on the relationship between the length and weight relationship of a given species (Fulton, 1902; Equation 5.1). Species such as *C. gobio* were treated with caution due to their variable length / weight ratios and limited resolution as many had a weight of less than 1 g.

$$K = \frac{W \cdot 100}{TL^3}$$

Equation 5.1.

Where condition factor (K), weight (W) is the weight of each individual (g) and length (L) is the length of each individual (cm).

5.2.3.3 Macroinvertebrate Density, Biomass and Community Structure

Invertebrate number and length were recorded for density and biomass calculations. As *Oligochaeta* spp. were often dismantled, sub-samples were used to generate estimates of species density and biomass. Reported length-weight regressions were used to determine biomass as ash free dry weight (AFDW) and scaled up to g m⁻² (Smock, 1980; Benke *et al.*, 1999; Stoffels *et al.*, 2003; Edwards *et al.*, 2009). A series of two-way ANOVAs on ranks were used to examine mean invertebrate density and biomass among sites. All statistics were performed using SigmaPlot 14.0. Summaries of dominant family contributions, taxa richness, Shannon-Weiner diversity (Krebs, 1989) and average score per taxon were examined for each site to determine differences in richness and communities between sites.

Ordinations of invertebrate community structure were examined using PRIMER (Clarke and Warwick, 2001). ANOSIM was used to examine similarity in mean density between restored and unrestored sites. Invertebrate community structure was log (x+1) transformed to normalise the data and examined using non-metric multidimensional scaling (NMDS) using measures of Bray-Curtis similarity. NMDS uses ordinations to

determine relationships between communities for density and biomass across reaches by ranking data on a site and species similarity matrix. Output values of between 0 and 1 are given, with a lower value highlighting a stronger matrix fit. Subsequently, SIMPER analysis was used to examine the percentage attribution of taxa density and biomass to differences observed between reaches (Clarke and Warwick, 2001). An additional SIMPER test was carried out to determine which taxa were driving similarities between sites.

5.3 Results

5.3.1 Physical Habitat

Assessments of physical habitats varied between rivers and reaches (Table 5.1). Widths ranged from 2.90 m to 6.88 m and differed significantly among rivers ($p < 0.001$). Overall widths did not differ between reaches, however site-specific examinations revealed significant differences between restored and unrestored reaches at the Pool, Wandle and Hogsmill (Table 5.1.), with increased widths at the restored Pool and Hogsmill sites. Depths ranged from 0.06 m to 0.40 m and showed significant differences between rivers ($F_{4,240} = 10.25$; $p < 0.001$) and reaches (restored and unrestored) ($F_{1,240} = 17.74$; $p < 0.001$). Upon closer inspection, Tukey's test revealed that this was primarily driven by shallower depths at the Pool and Hogsmill restored sites (Table 5.1). Velocity ranged from 0.14 m s⁻¹ to 0.38 m s⁻¹ at restored sites to 0.01 m s⁻¹ to 0.37 m s⁻¹ at unrestored sites. Velocity was significantly different between rivers but when all restored reaches were compared against unrestored reaches, there was no significant difference. Despite this, Tukey's test revealed site-specific differences between reaches at the Hogsmill and Brent, however these display contradictory results in terms of velocity response.

Table 5.1. A summary of hydromorphological characteristics across sites for width ($n = 5$), depth ($n = 25$) and velocity ($n = 25$). Values in parentheses denote one standard error. Significant differences between restored and unrestored reaches are in bold; differences among rivers are indicated by letter groupings. Overall, width, depth and velocity differed significantly between sites. However, depth was the only parameter that responded significantly to restoration, exhibiting shallower depths at restored sites ($F_{4,240} = 17.74$; $p < 0.001$).

River	Reach	Width (m)	Depth (m)	Velocity (m s^{-1})
Ravensbourne	Restored	6.70 ^a (0.21)	0.26 ^a (0.02)	0.17 ^c (0.02)
	Unrestored	6.88 ^a (0.20)	0.33 ^a (0.03)	0.23 ^c (0.01)
Pool	Restored	6.11^{a,d} (0.21)	0.18^d (0.02)	0.14 ^b (0.01)
	Unrestored	6.82^{a,d} (0.04)	0.40^d (0.02)	0.01 ^b (0.003)
Wandle	Restored	2.90^b (0.14)	0.22 ^{c,e,b} (0.02)	0.29 ^{d,c} (0.04)
	Unrestored	3.74^b (0.12)	0.17 ^{c,e,b} (0.01)	0.24 ^{d,c} (0.03)
Hogsmill	Restored	4.11^c (0.10)	0.06^b (0.01)	0.38^a (0.04)
	Unrestored	3.30^c (0.06)	0.25^b (0.05)	0.12^a (0.02)
Brent	Restored	6.19 ^d (0.67)	0.28 ^{a,e} (0.05)	0.18^a (0.03)
	Unrestored	6.03 ^d (0.55)	0.21 ^{a,e} (0.02)	0.37^a (0.03)

5.3.2 Fish

5.3.2.1 Fish Abundance

A total of nine fish species (comprising 1541 individuals) were recorded in this study; 955 at restored sites and 586 at unrestored sites (Figure 5.1-5.11). The highest estimates of abundance were recorded for both the Hogsmill and Wandle restored sites (Hogsmill 566; Wandle 471, Figure 5.1; Table 5.2). The Wandle and Hogsmill unrestored sites also surpassed abundance numbers across all other sites (Wandle 425; Hogsmill 191). These results are primarily attributed to a dominance of *C. gobio*, which represented 53% and 71% of the total number of individuals caught across all sites at restored and unrestored sites, respectively (Figure 5.1). Common minnow *Phoxinus phoxinus* was the second most abundant species at restored sites, accounting for 19% of the estimated abundance, with stickleback *Gasterosteus aculeatus* representing the second largest group of species at unrestored sites (16%). The lowest abundance estimates were recorded at the restored Brent site, where a single *A. anguilla* was caught amongst an estimated 13 *G. aculeatus*. By comparison, the highest number of *A. anguilla* (18) was recorded at the paired unrestored reach. A total of 13 *S. trutta* individuals were detected at the Wandle alone, with 12 of these recorded at the restored site (Figure 5.1; Table 5.2). *R. rutilus* and *P. fluviatilis* were also detected in isolation at the Pool unrestored site. The greatest depths were also recorded at this site (0.40 m). Catch efficiencies varied among species caught (Table 5.3) with the highest values being observed for *L. leuciscus* (0.78), *S. trutta* (0.75) and *A. anguilla* (0.67-0.75). The lowest values were detected for *C. gobio* (0.27-0.39), *P. phoxinus* (0.44) and *G. aculeatus* (0.26-0.65).

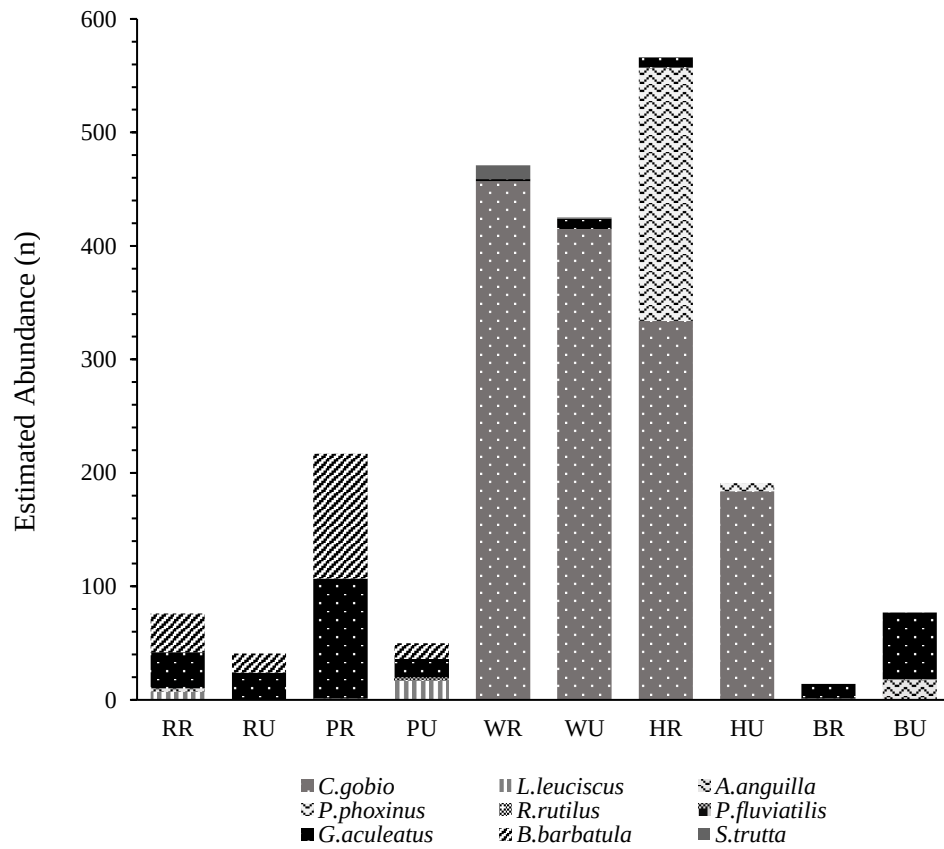


Figure 5.1. Estimated fish abundance (n) per site, for all sites combined. Separate bars within columns denote different species. Estimated fish abundance ranged from 14 (Brent restored) to 566 (Hogsmill restored). *C. gobio* contributed largely to the high number of fish caught.

Table 5.2. A summary of estimated fish abundance incorporating all individuals caught across three 25 metre sweeps. Values not in bold denote estimated fish abundance; values in bold denote total sum where depletion did not occur. Values in parentheses denote variance. Dashed lines denote an absence of individuals.

Species	Ravensbourne		Pool		Wandle		Hogsmill		Brent	
	Rest.	Unrest.	Rest.	Unrest.	Rest.	Unrest.	Rest.	Unrest.	Rest.	Unrest.
<i>C. gobio</i>	-	-	-	-	457 (328-586)	415 (311-519)	334 (250-418)	184 (147-221)	-	-
<i>L. leuciscus</i>	7 (6-7)	-	-	17	-	-	-	-	-	-
<i>A. anguilla</i>	3 (2-4)	-	1	-	-	-	-	-	1	18 (16-20)
<i>P. phoxinus</i>	-	-	-	-	-	-	223 (195-253)	7	-	-
<i>R. rutilus</i>	-	-	-	1	-	-	-	-	-	-
<i>P. fluviatilis</i>	-	-	-	2	-	-	-	-	-	-
<i>G. aculeatus</i>	32 (21-43)	24 (18-30)	106 (38-174)	16	2	9	9	-	13 (11-15)	59 (43-75)
<i>B. barbatula</i>	34 (12-56)	17 (13-21)	110 (95-125)	14 (12-16)	-	-	-	-	-	-
<i>S. trutta</i>	-	-	-	-	12 (11-13)	1	-	-	-	-
Total	76	41	217	50	471	425	566	191	14	77

Table 5.3. A summary of catch efficiency values for species examined used the Carle and Strub Method. Values closer to one denote a higher catch efficiency. Values reflect variability among species caught). Dashed lines denote where no fish were caught. The lowest values were detected for *C. gobio* (0.27-0.39), *P. phoxinus* (0.44) and *G. aculeatus* (0.26-0.65), while the highest efficiencies were seen among *L. leuciscus* (0.78), *S. trutta* (0.75) and *A. anguilla* (0.67-0.75).

Species	Ravensbourne		Pool		Wandle		Hogsmill		Brent	
	Rest.	Unrest.	Rest.	Unrest.	Rest.	Unrest.	Rest.	Unrest.	Rest.	Unrest.
<i>C. gobio</i>	-	-	-	-	0.27	0.30	0.31	0.39	-	-
<i>L. leuciscus</i>	0.78	-	-	-	-	-	-	-	-	-
<i>A. anguilla</i>	0.75	-	-	-	-	-	-	-	-	0.67
<i>P. phoxinus</i>	-	-	-	-	-	-	0.44	-	-	-
<i>R. rutilus</i>	-	-	-	-	-	-	-	-	-	-
<i>P. fluviatilis</i>	-	-	-	-	-	-	-	-	-	-
<i>G. aculeatus</i>	0.44	0.54	0.26	-	-	-	-	-	0.65	0.44
<i>B. barbatula</i>	0.34	0.55	0.50	0.7	-	-	-	-	-	-
<i>S. trutta</i>	-	-	-	-	0.75	-	-	-	-	-

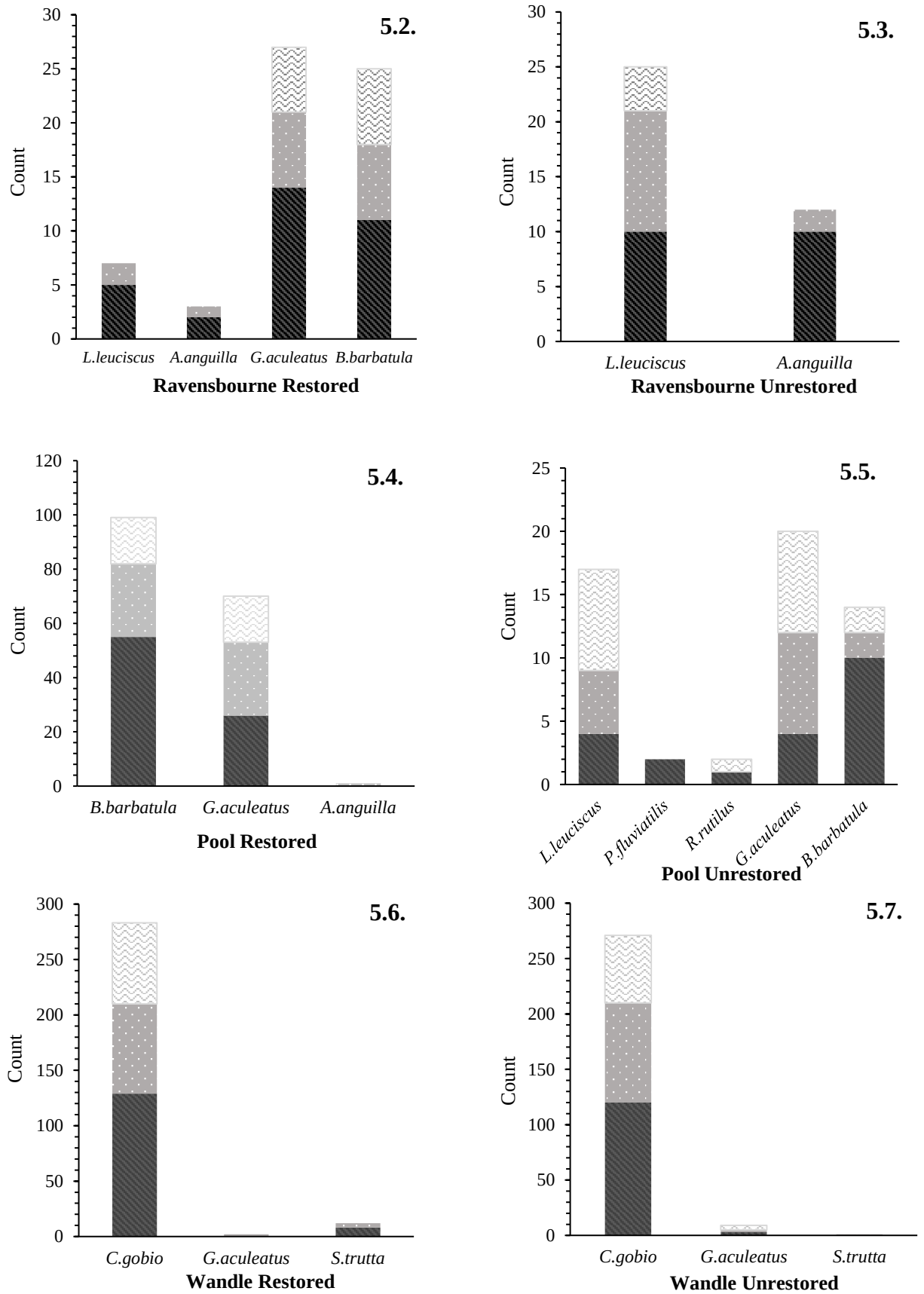


Figure 5.2-5.7. Fish count for each sweep per site. Bars denote fish number caught in each sweep, in ascending order (1-3): = sweep 1, = sweep 2, = sweep 3.

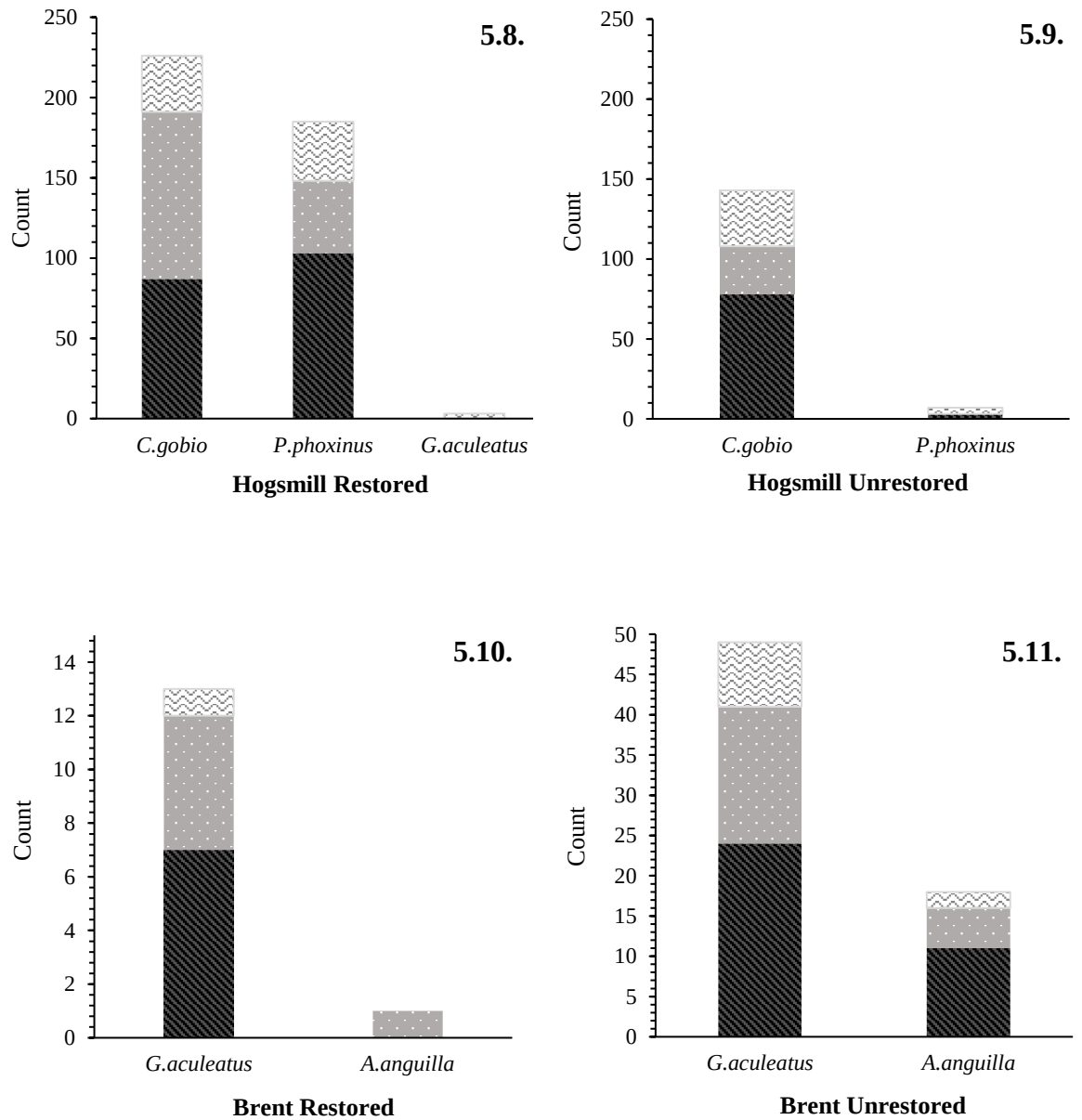


Figure 5.8-5.11. Fish count for each sweep per site. Bars denote fish number caught in each sweep, in ascending order (1-3):  = sweep 1,  = sweep 2,  = sweep 3.

5.3.2.2 Fish Biomass

A total estimated wet biomass of 8021 g was recorded across all sites; 5072 g at restored sites and 2949 g at unrestored sites (Table 5.4; Figure 5.12). Biomass for reaches differed with the lowest values recorded at the Brent restored reach (26 g) and the highest value was recorded at the Wandle restored reach (2746 g). Despite significant numbers of *C. gobio*, elevated biomass at the restored site was driven by the presence of 12 adult *S. trutta*, accounting for a total mass of 2155 g (Table 5.4). The restored River Wandle reach also had the highest abundance of *C. gobio* (Table 5.2), although biomass numbers for species at this site were markedly lower than that of the paired unrestored site. The Pool and Brent were the only rivers where overall biomass values were greater at unrestored sites when compared with restored sites, driven by *L. leuciscus* (999 g) and *A. Anguilla* (153 g), respectively (Figure 5.12). Fulton's condition factor measured for *S. trutta* at the Wandle revealed a mean value of 1.28 ($\pm$ 0.57) at the restored site and 1.01 for the only fish sampled at the unrestored site.

Table 5.4. A summary of total fish biomass (g), incorporating all individuals caught across three 25 metre sweeps. Dashes denote an absence of species biomass.

Species	Ravensbourne		Pool		Wandle		Hogsmill		Brent	
	Rest.	Unrest.	Rest.	Unrest.	Rest.	Unrest.	Rest.	Unrest.	Rest.	Unrest.
<i>C. gobio</i>	0	0	0	0	587	1000	631	388	-	-
<i>L. leuciscus</i>	95	-	-	999	-	-	-	-	-	-
<i>A. anguilla</i>	123	-	58	-	-	-	-	-	2	153
<i>P. phoxinus</i>	-	-	-	-	-	-	747	24	-	-
<i>R. rutilus</i>	-	-	-	2	-	-	-	-	-	-
<i>P. fluviatilis</i>	-	-	-	31	-	-	-	-	-	-
<i>G. aculeatus</i>	28	20	62	15	4	24	8	-	24	70
<i>B. barbatula</i>	137	53	411	90	-	-	-	-	-	-
<i>S. trutta</i>	-	-	-	-	2155	80	-	-	-	-
Total	383	73	531	1137	2746	1104	1386	412	26	223

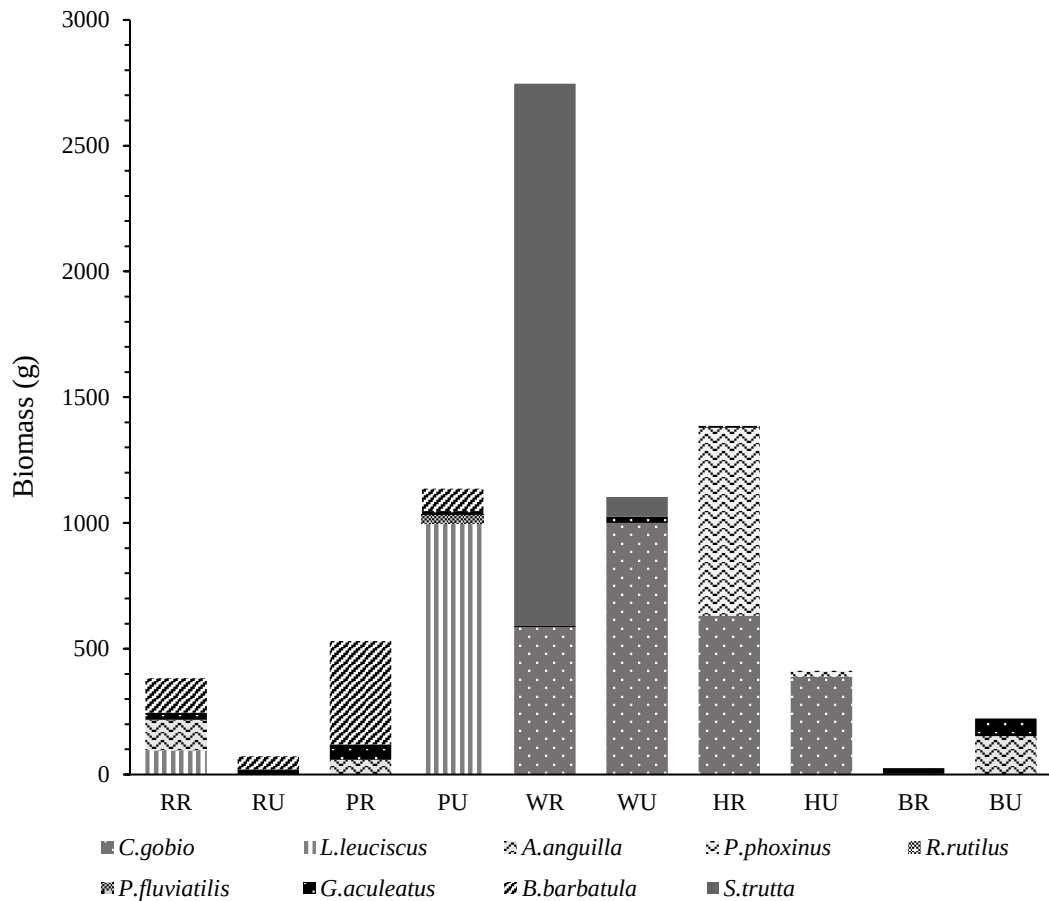


Figure 5.12. Total fish biomass (g) per site, separated by species.

5.3.3 Macroinvertebrates

5.3.3.1 Macroinvertebrate Density and Richness

A total of 5,444 invertebrate individuals comprising 25 taxa were identified across all sites. 23 families representing orders Trichoptera, Isopoda, Diptera, Ephemeroptera, Tricladida, Coleoptera, Ryhnchondellida, Arhynchobdellida, Amphipoda, Littorinimorpha, Basommatophora and Oligochaeta were recorded. No invasive species were observed in samples. Mean invertebrate density ranged from 138 to 2130 individuals per m⁻², with the highest and lowest values being observed at the Wandle restored and Hogsmill unrestored sites, respectively. Density varied significantly across rivers sites ($F_{4,40} = 6.61$; $p < 0.01$), but not between reaches ($F_{1,40} = 3.71$; $p = 0.06$). Between sites, the Wandle and Brent showed no difference in density, although this similarity was primarily driven by Oligochaeta spp. at the Brent and *G. pulex* at the Wandle (Table 5.5).

Table 5.5. A summary of mean invertebrate density (n per m^2) for five surber samples across paired restored and unrestored reaches. Values in parentheses denote one standard error.

Species	Ravensbourne		Pool		Wandle		Hogsmill		Brent	
	Rest.	Unrest.	Rest.	Unrest.	Rest.	Unrest.	Rest.	Unrest.	Rest.	Unrest.
<i>A. fuscipes</i>	2 (2)	-	13 (11)	-	-	39 (34)	55 (29)	-	-	-
<i>A. aquaticus</i>	125 (34)	147 (36)	174 (102)	46 (9)	687 (399)	301 (283)	42 (5)	6 (2)	749 (582)	532 (73)
Baetis spp.	2 (2)	-	9 (9)	-	42 (23)	44 (22)	46 (20)	7 (7)	-	-
Ceratopogonidae spp.	-	-	-	-	4 (4)	7 (4)	-	-	-	-
Chironomidae spp.	72 (39)	51 (18)	26 (7)	9 (9)	147 (69)	20 (10)	147 (73)	9 (5)	13 (5)	20 (9)
<i>D. lacteum</i>	-	4 (4)	4 (4)	-	6 (6)	4 (4)	-	-	-	-
<i>E. aenea</i>	-	-	-	-	13 (5)	26 (11)	7 (3)	2 (2)	-	-
<i>E. octoculata</i>	7 (4)	13 (11)	11 (9)	7 (7)	17 (12)	24 (17)	4 (2)	-	29 (11)	13 (5)
<i>E. testacea</i>	20 (11)	31 (14)	15 (4)	4 (2)	26 (12)	7 (2)	2 (2)	-	68 (13)	81 (43)
<i>G. pulex</i>	44 (13)	42 (18)	499 (167)	17 (5)	799 (265)	1155 (367)	296 (70)	51 (11)	106 (82)	48 (18)
<i>G. complanata</i>	2 (2)	2 (2)	7 (7)	-	40 (31)	7 (3)	13 (11)	-	2 (2)	7 (4)
<i>H. stagnalis</i>	-	11 (5)	-	-	26 (9)	-	2 (2)	-	180 (66)	118 (61)
Hydrobiidae spp.	-	-	2 (2)	-	-	-	-	-	-	-

Table 5.5 Continued. A summary of mean invertebrate density for five surber samples (n per m^2) across paired restored and unrestored reaches. Values in parentheses denote one standard error.

Species	Ravensbourne		Pool		Wandle		Hogsmill		Brent	
	Rest.	Unrest.	Rest.	Unrest.	Rest.	Unrest.	Rest.	Unrest.	Rest.	Unrest.
<i>L. mysticoides</i>	28 (15)	35 (13)	4 (2)	4 (2)	-	-	-	-	-	-
Limnephilidae spp.	-	-	-	-	44 (24)	4 (2)	-	-	-	-
Muscidae spp.	2 (2)	6 (6)	-	-	6 (4)	4 (4)	2 (2)	-	-	-
Physidae spp.	-	-	-	-	11 (7)	-	-	4 (2)	33 (33)	81 (43)
Planorbidae spp.	4 (2)	2 (2)	-	(2 (2)	17 (12)	4 (4)	9 (4)	-	-	4 (4)
Polycelis spp.	6 (2)	9 (3)	33 (11)	4 (2)	77 (24)	4 (2)	4 (4)	-	11 (3)	26 (13)
Psychomyiidae spp.	-	-	-	-	4 (4)	-	-	-	-	-
Rhyacophila spp.	-	-	-	-	29 (9)	39 (14)	-	-	-	-
<i>S. ignita</i>	-	-	-	-	11 (5)	11 (7)	-	-	-	-
Simulium spp.	-	-	-	-	81(43)	-	4 (4)	-	-	-
Zygoptera spp.	-	-	-	-	-	-	-	2 (2)	-	-
Oligochaeta spp.	73 (19)	110 (33)	118 (32)	64 (19)	46(15)	72 (8)	162 (67)	57 (35)	804 (476)	321 (68)
Total	386 ^{a,b}	463 ^{a,b}	914 ^{d,b}	156 ^{d,b}	2130 ^b	1770 ^b	793 ^{c,b}	138 ^{c,b}	1996 ^{b,b}	1250 ^b
	(98)	(76)	(184)	(30)	(532)	(245)	(197)	(35)	(1065)	(192)

G. pulex was the dominant species at restored sites, contributing on average 22.93% (± 0.09) to total taxa. *G. pulex* contributed 54.54%, 37.51% and 65.25% to total density at the Pool, Wandle and Hogsmill, respectively. Similarly, at unrestored sites, *G. pulex* was highly dominant, contributing to 31.90% of the total mean density (± 0.18). *A. aquaticus* was the second most abundant species, accounting for 26.30% (mean) and across sites 31.70%, 41.02%, 41.3% and 42.56% of total density at the Ravensbourne, Pool, Hogsmill and Brent.

Mean taxa richness ranged from 4 to 14 (Table 5.6), revealing significant differences between rivers ($F_{4,40} = 24.77$; $p < 0.001$) and reaches ($F_{1,40} = 20.21$; $p < 0.001$). Tukey's test revealed that these differences were driven by reach differences at the Pool, Wandle and Hogsmill. Further examination revealed that interactions between rivers and reaches were significant ($F_{4,40} = 8.94$; $p < 0.001$), predominantly driven by the exclusive detection of *S. ignita* and Limnephilidae spp. at the Wandle. Average Score Per Taxon (ASPT) values ranged from 3.19 to 4.58 (Table 5.6) and differed between rivers ($F_{4,40} = 4.67$; $p = 0.003$), but not reaches ($F_{1,40} = 0.10$; $p = 0.76$). Mean Shannon-Weiner index values ranged from 0.86 to 1.69 (Table 5.6) and varied significantly between rivers ($F_{4,40} = 6.17$; $p < 0.001$) and reaches ($F_{1,40} = 11.83$; $p = 0.001$). Tukey's test revealed that these differences were driven by significantly higher values at the Wandle and Hogsmill restored sites ($p < 0.001$).

Table 5.6. A summary of taxa richness (*n*), average score per taxon (ASPT) and Shannon-Weiner (SW) indexes across rivers and reaches. Significant differences between rivers are indicated by letter groupings.

Indices	Ravensbourne		Pool		Wandle		Hogsmill		Brent	
	Rest.	Unrest.	Rest.	Unrest.	Rest.	Unrest.	Rest.	Unrest.	Rest.	Unrest.
Richness	7 ^{c,d,b}	9 ^{c,d,b}	8 ^b	5 ^b	14 ^a	10 ^a	9 ^{b,d}	4 ^{b,d}	8 ^d	9 ^d
	(1)	(1)	(0.5)	(1)	(1)	(1)	(1)	(0.3)	(1)	(0.40)
ASPT	3.8 ^a	4.58 ^a	3.88 ^a	3.86 ^a	4.49 ^a	4.44 ^a	3.84 ^a	3.52 ^a	3.27 ^{b,a}	3.19 ^{b,a}
	(0.56)	(0.42)	(0.29)	(0.35)	(0.33)	(0.24)	(0.09)	(0.26)	(0.08)	(0.09)
SW	1.61 ^a	1.69 ^a	1.21 ^{b,c}	1.28 ^{b,c}	1.64 ^{c,e}	0.86 ^{c,e}	1.57 ^{d,e}	1.0 ^{d,e}	1.36 ^e	1.49 ^e
	(0.06)	(0.11)	(0.13)	(0.11)	(0.13)	(0.09)	(0.10)	(0.05)	(0.10)	(0.08)

5.3.3.2 Macroinvertebrate Biomass

Mean biomass (ash free dry weight) throughout all sites ranged from 0.17 to 5.63 g m⁻², with restored sites exhibiting higher mean biomass values combined (13.40 g) than unrestored sites (5.89 g) (Table 5.7). Tukey's test revealed that biomass differed between rivers ($F_{4,40} = 6.93$; $p < 0.001$), with significantly higher values at restored reaches ($F_{1,40} = 10.47$; $p = 0.002$). Closer inspection highlighted that this was driven by higher values at the Wandle (5.63 g m⁻²) and Hogsmill (2.55 g m⁻²). By comparison, unrestored sites exhibited the lowest biomass values (Pool: 0.17 g m⁻², Hogsmill: 0.24 g m⁻²), with both sites collectively contributing only 2.13% to the total biomass across all sites. At restored sites, *G. pulex* contributed 38%, 79% and 63% of total mean biomass at the Ravensbourne, Pool and Wandle, respectively, with Planorbidae spp. (51%) and *E. testacea* contributing 29% to the total biomass at the Hogsmill and Brent. At unrestored sites, *G. pulex* was the dominant species accounting for biomass at the Pool (49%), Wandle (66%) and Hogsmill (75%), with the Ravensbourne dominated by *E. octoculata* (29%) and the Brent *E. testacea* (45%) exhibiting higher collective biomass values (restored mean total: 13.40 g; unrestored mean total: 5.89 g (Table 5.7)

Table 5.7. A summary of mean invertebrate biomass for five surber samples (g per m⁻²) across paired restored and unrestored reaches (continued next page).

Species	Ravensbourne		Pool		Wandle		Hogsmill		Brent	
	Rest.	Unrest.	Rest.	Unrest.	Rest.	Unrest.	Rest.	Unrest.	Rest.	Unrest.
<i>A. fuscipes</i>	<0.001 (<0.001)	-	0.002 (0.002)	-	-	0.001 (0.001)	0.01 (0.003)	-	-	-
<i>A. aquaticus</i>	0.05 (0.02)	0.05 (0.02)	0.03 (0.02)	0.02 (0.005)	0.19 (0.04)	0.18 (0.18)	0.01 (0.002)	0.004 (0.003)	<0.001 (<0.001)	<0.001 (<0.001)
Baetis spp.	0.002 (0.002)	-	0.21 (0.21)	-	0.43 (0.28)	0.23 (0.13)	0.18 (0.07)	0.03 (0.03)	-	-
Ceratopogonidae spp.	-	-	-	-	0.01 (0.01)	0.01 (0.01)	-	-	-	-
Chironomidae spp.	0.02 (0.01)	0.03 (0.02)	0.01 (0.003)	0.01 (0.01)	0.03 (0.01)	0.002 (0.001)	0.22 (0.13)	0.004 (0.002)	0.01 (0.005)	0.02 (0.01)
<i>D. lacteum</i>	-	0.01 (0.01)	0.01 (0.01)	-	0.02 (0.02)	0.01 (0.01)	-	-	-	-
<i>E. aenea</i>	-	-	-	-	0.004 (0.002)	0.002 (0.001)	0.003 (0.001)	<0.001 (<0.001)	-	-

Table 5.7. Continued. A summary of mean invertebrate biomass for five surber samples (g per m⁻²) across paired restored and unrestored reaches.

Species	Ravensbourne		Pool		Wandle		Hogsmill		Brent	
	Rest.	Unrest.	Rest.	Unrest.	Rest.	Unrest.	Rest.	Unrest.	Rest.	Unrest.
<i>E. octoculata</i>	0.02 (0.01)	0.17 (0.17)	0.02 (0.02)	0.01 (0.01)	0.17 (0.13)	0.06 (0.03)	0.003 (0.002)	-	0.03 (0.01)	0.01 (0.01)
<i>E. testacea</i>	0.09 (0.06)	0.14 (0.01)	0.03 (0.01)	0.004 (0.003)	0.01 (0.006)	0.03 (0.01)	0.02 (0.02)	-	0.81 (0.24)	1.27 (0.55)
<i>G. pulex</i>	0.16 (0.06)	0.09 (0.02)	1.56 (0.46)	0.08 (0.03)	3.56 (1.47)	1.35 (0.61)	1.0 (0.2)	0.18 (0.04)	0.32 (0.14)	0.35 (0.16)
<i>G. complanata</i>	0.003 (0.002)	0.008 (0.008)	0.02 (0.02)	-	0.04 (0.03)	0.02 (0.01)	0.03 (0.02)	-	0.02 (10.02)	0.06 (0.05)
<i>H. stagnalis</i>	-	0.02 (0.01)	-	-	0.05 (0.02)	-	0.01 (0.009)	-	0.80 (0.31)	0.43 (0.28)
Hydrobiidae spp.	-	-	0.06 (0.06)	-	-	-	-	-	-	-

Table 5.7. Continued. A summary of mean invertebrate biomass for five surber samples (g per m⁻²) across paired restored and unrestored reaches.

Species	Ravensbourne		Pool		Wandle		Hogsmill		Brent	
	Rest.	Unrest.	Rest.	Unrest.	Rest.	Unrest.	Rest.	Unrest.	Rest.	Unrest.
<i>L. mysticides</i>	0.009 (0.005)	0.006 (0.003)	0.75 (0.52)	0.56 (0.36)	-	-	-	-	-	-
Limnephilidae spp.	-	-	-	-	0.58 (0.31)	0.009 (0.006)	-	-	-	-
Muscidae spp.	0.003 (0.003)	0.006 (0.006)	-	-	0.003 (0.002)	0.001 (0.001)	0.001 (0.001)	-	-	-
Physidae spp.	-	-	-	-	0.022 (0.017)	-	-	0.01 (0.01)	0.07 (0.07)	0.11 (0.1)
Planorbidae spp.	0.04 (0.03)	0.05 (0.05)	-	0.03 (0.03)	0.38 (0.38)	0.06 (0.06)	1.06 (0.71)	-	-	0.16 (0.16)
Polycelis spp.	<0.001 (<0.001)	0.002 (0.71)	0.005 (0.001)	<0.001 (<0.001)	0.007 (0.002)	<0.001 (<0.001)	<0.001 (<0.001)	-	0.002 (0.84)	0.004 (0.002)

Table 5.7. Continued: A summary of mean invertebrate biomass for five surber samples (g per m⁻²) across paired restored and unrestored reaches.

Species	Ravensbourne		Pool		Wandle		Hogsmill		Brent	
	Rest.	Unrest.	Rest.	Unrest.	Rest.	Unrest.	Rest.	Unrest.	Rest.	Unrest.
Psychomyiidae spp.	-	-	-	-	0.01 (0.01)	-	-	-	-	-
Rhyacophila spp.	-	-	-	-	0.10 (0.04)	0.05 (0.04)	-	-	-	-
<i>S. ignita</i>	-	-	-	-	0.01 (0.01)	0.003 (2.53)	-	-	-	-
Simulium spp.	-	-	-	-	<0.001 (<0.01)	-	<0.001 (<0.01)	-	-	-
Zygoptera spp.	-	-	-	-	-	-	-	0.01 (0.01)	-	-
Oligochaeta spp.	0.01 (0.02)	0.02 (0.01)	0.03 (0.01)	0.02 (0.01)	0.01 (0.01)	0.01 (0.005)	0.02 (0.01)	0.01 (0.003)	0.17 (0.09)	0.09 (0.03)
Total	0.43 (0.07)	0.61 (0.19)	1.97 (0.61)	0.17 (0.05)	5.63 (1.59)	2.06 (0.67)	2.55 (0.83)	0.24 (0.04)	2.82 (0.83)	2.82 (0.78)

5.3.3.3 Macroinvertebrate Community Ordinations

ANOSIM results for density revealed differences across all river groups ($R = 0.62$), with hierarchical cluster analysis depicting notable groupings within the Brent, Wandle (restored and unrestored), Hogsmill and Ravensbourne. Non-metric multidimensional scaling (NMDS) plots detected marginally significant clustering among sites (stress = 0.18) (Figure 5.13). Both reaches of the Brent demonstrated the closest cluster of results, where the highest density of *Erpobdella* spp. and *Oligochaeta* spp. were found. Generally, sites with higher invertebrate densities (Wandle and Brent) were positioned to the left of the plot, showing a gradual gradient drop alongside density to the right of the plot (Hogsmill and Pool unrestored sites). The positioning of the Hogsmill unrestored site is less conclusive, illustrating the lowest degree of clustering when compared to the other sites.

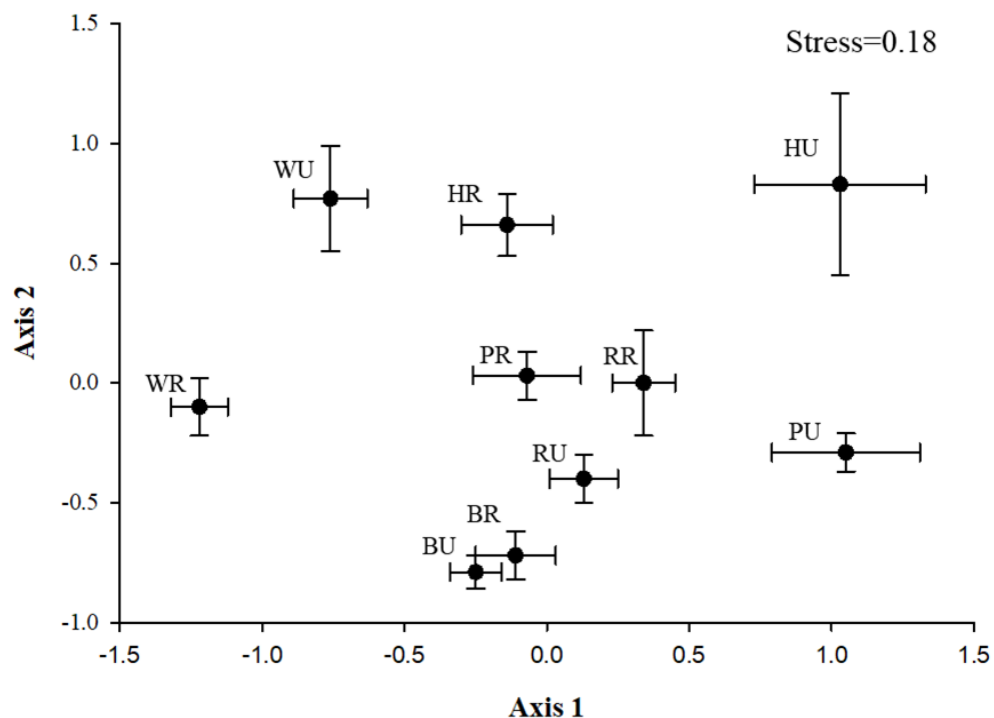


Figure 5.13. Non-metric multidimensional scaling (NMDS) using measures of Bray-Curtis similarity, highlighting community clustering between sites.

SIMPER analysis revealed 39.30% dissimilarity in mean species density between restored and unrestored sites (Table 5.8). Chironomidae spp. contributed the most to this value (10.25% dissimilarity), with *G. pulex*, *A. aquaticus* and Polycelis spp. also contributing similar degrees of influence upon dissimilarity (8.14-8.5%). A second SIMPER analysis examining similarity between paired reaches displayed the highest degree of similarity at the Brent (76.30), driven by *A. aquaticus* (21.35%), Oligochaeta (20.28%) and *A. aquaticus* (14.13%). NMDS plots for biomass revealed no patterns in distribution (Stress = 0.2).

Table 5.8. Results of a SIMPER analysis to determine the contribution of important taxa to mean dissimilarity of density between restored and unrestored sites. Data showing the top 12 most abundant taxa.

Test Groups	Mean Dissimilarity (%)	Taxa	Dissimilarity (%)	Cumulative Dissimilarity (%)
Restored vs. Unrestored	39.30	Chironomidae spp.	10.25	10.25
		<i>G. pulex</i>	8.50	18.75
		<i>A. aquaticus</i>	8.30	27.05
		Polycelis spp	8.14	35.19
		<i>A. fuscipes</i>	6.78	41.97
		<i>E. octoculata</i>	6.32	48.29
		<i>E. testacea</i>	5.92	54.21
		Baetidae spp.	5.50	59.71
		Oligochaeta spp.	5.52	65.12
		<i>H. stagnalis</i>	4.77	69.90
		Planorbidae spp.	4.60	74.49
		<i>G. complanata</i>	4.44	78.93

Table 5.9. Results highlighting contributions of taxa to mean similarity between paired reaches, based on the top 5 most abundant taxa.

Test Groups	Mean Similarity (%)	Taxa	Similarity (%)	Cumulative Similarity (%)
Ravensbourne	68.68	<i>A. aquaticus</i>	23.11	23.11
		Oligochaeta spp.	21.02	44.13
		<i>G. pulex</i>	15.89	60.02
		Chironomidae spp.	13.73	73.75
		<i>L. mysticides</i>	8.98	82.73
Pool	64.01	Oligochaeta spp.	30.97	30.97
		<i>A. aquaticus</i>	24.77	24.77
		<i>G. pulex</i>	19.66	75.39
		<i>E. testacea</i>	7.65	83.04
		Chironomidae spp.	7.51	90.56
Wandle	60.89	<i>G. pulex</i>	20.33	20.33
		Oligochaeta spp.	12.41	32.74
		<i>A. aquaticus</i>	11.85	44.59
		Chironomidae spp.	7.66	52.25
		<i>R. dorsalis</i>	7.31	59.56
Hogsmill	63.26	<i>G. pulex</i>	35.77	35.77
		Oligochaeta spp.	22.23	58.00
		<i>A. aquaticus</i>	12.77	70.76
		Chironomidae spp.	11.83	82.59
		<i>A. fuscipes</i>	6.17	88.76
Brent	76.30	<i>A. aquaticus</i>	21.35	21.35
		Oligochaeta spp.	20.28	41.63
		<i>E. testacea</i>	14.13	55.76
		<i>H. stagnalis</i>	12.56	68.32
		<i>G. pulex</i>	11.10	79.42

5.4 Discussion

This study aimed to deliver a quantifiable assessment of fish and invertebrate community structure as an indicator of overall stream health. This was achieved through an assessment of fish communities, macroinvertebrate community structure and broad physical habitat measurements. It was hypothesised that **(1)** fish abundance, biomass and richness would increase at restored sites due to improvements in hydromorphological conditions fostering ecological integrity, **(2)** Macroinvertebrate density, biomass and richness would increase at restored sites due to improvements in water and habitat quality

and **(3)** Physical habitat heterogeneity would increase at restored sites due to improvements in hydrological heterogeneity and habitat diversity.

In this study, overall fish abundance increased at restored sites (Hypothesis 1), largely due to the presence of *C. gobio* (i.e. Wandle and Hogsmill; Figure 5.1; Table 5.2). *C. gobio* are associated with good water quality and favour environments characterised by coarse substrata and heterogeneous flows (Pretty *et al.*, 2003; Carter and England, 2004a). *C. gobio* can also withstand infrequent pollution episodes, as previously noted in the Great Ouse (Copp *et al.*, 1994). Studies carried out by the Environment Agency examining fish in London rivers between 1991 and 2003 reported a total of 29 fish species across 91 sites (Carter and England, 2004a). These included marine and migratory species which occur in both freshwater and intertidal reaches (Carter and England, 2004a). *A. anguilla* and *R. rutilus* were the two most abundant species recorded. Conversely, *A. anguilla* was recorded at only four sites in this study, with notable increases in abundance at the Brent unrestored site where the artificial passage and walls may have provided shelter.

At the River Wandle, *S. trutta* increased in abundance and biomass at restored site (Hypothesis 1). 12 individuals of *S. trutta* were counted at the restored Wandle site, which differs from previous work at this river where no counts were made (Carter and England, 2004a). Stocking has significantly influenced *S. trutta* abundance along this reach by subsidising species numbers and providing a mechanism for dispersal. The mean condition factor of *S. trutta* found that the Wandle restored reach (1.1) fell within previously reported values (1.3 ± 0.11) (Rechulicz, 2010). Condition factors can be influenced by seasonality (with spring yielding higher values), food availability, temperature, pH and genotype (Frost and Brown, 1967). Hard alkaline waters have been characterised by lower populations and increased food supplies, while acidic waters have been found to suppress growth through reductions in food availability and spawning

grounds (Frost and Brown, 1967). Other work has shown that chalk stream rivers exhibiting warm temperatures can promote increased growth rates of *S. trutta* (Elliott *et al.*, 1975). In this study, flow was not found to be a significant factor influencing change (Hypothesis 3). However, it should be acknowledged that the results taken from this single survey may not necessarily represent longer temporal conditions.

Catch efficiencies for some several species assessed in this study (e.g. *C. gobio*) were low. This is likely due to their cryptic nature, ability to utilise sheltered habitats (e.g. under rocks or in the bankside) and discrete colouration (Harrison *et al.*, 2005). In addition, electric fishing is often targeted at salmonids as other species do not respond to currents as effectively. Baldwin and Aprahamian (2012) highlighted the difficulties associated with attaining accurate population estimates for eels. Eels are known to be misrepresented in electric fishing catches due to their size and poor attraction to the anode.

Macroinvertebrate biomass and richness varied significantly across habitat patches in this study. *G. pulex*, a highly tolerant and generalist species favouring decomposing leaf litter, detritus and algae (Ledger *et al.*, 2006), was present at all restored sites. Resistant taxa *Erpobdella* spp. and *Oligochaeta* spp. were highly abundant at the Brent, possibly attributed to elevated concentrations of ammonium (NH_4^+) previously detected at this site (Chapter 3). This shows similarities with other studies highlighting their presence at high concentrations, and an increase in organic and inorganic nutrients associated with increases in density and decreases in richness of macroinvertebrates (Pascoal *et al.*, 2005). Macroinvertebrate community structure is known to be influenced by sewage from stormwater tanks. Furthermore, flashy floods observed in urban environments can jeopardise the habitats needed to support invertebrates by overturning substrata and uprooting macrophytes. Post-disturbance conditions may reflect the resilience of species and the interactions between colonists and fauna (Ledger *et al.*, 2006). In mesocosm

experiments mimicking the effects of disturbance events on species of chalk streams, snails were found to engineer favourable habitats for filter feeders (Simuliidae spp.) and invertebrate predators, while deterring non-predatory Chironomids (Ledger *et al.*, 2006). In contrast, *G. pulex* increased in density (665%) due to rapid reproduction and *A. aquaticus* decreased as a result.

The Wandle had the highest levels of species richness, with increased numbers of *S. ignita* at restored reaches (Hypothesis 3). *S. ignita* are associated with good water quality, complex vegetation structures and fast-flowing waters (Carter and England, 2004b). They favour algae attached to stream features and detritus present at these sites. Studies have also found that poor oxygenation can also affect site occupancy in rivers for *S. ignita*, which play a vital role in aquatic and riparian food webs (Verberk *et al.*, 2016). Previous work has noted their absence at urban gradients > 20% (Stranko *et al.*, 2008; Stranko *et al.*, 2012). Increased abundances may be attributed to aeration of sediment and water provided by upstream water features. The in-channel characteristics arising from restoration work at the Wandle also provide a possible explanation for this, where macrophytes may have enabled the trapping of fine sediment, providing a vital zone for collecting and feeding of fine particle detritus (Verberk *et al.*, 2016).

ASPT values ranged from 3.19 to 4.58. At some of the sites where higher ASPT values might be expected, for example the Wandle, lower values were observed. This could be due to several factors including regional variations in ASPT and patchiness associated with natural variation. The dominance of *A. aquaticus* and *G. pulex* also points to the limited potential for restoration across some of these reaches which are already heavily degraded. The Brent restored site was heavily dominated by Oligochaeta spp., which may be attributed to high pollution loads of NH_4^+ and high biomass of tolerant fish species using this taxa as a food source.

There are numerous reasons why invertebrates may not be responding to restoration efforts. Foremost, the function of riffle-pools and flow deflectors at restored sites may not differ to that of reference sites. The location of these features in lowland urban environments is less likely to deliver the same benefits to that of less disturbed uplands. In upland environments, natural riffle-pools occur as a result of complex interactions between large-scale hydraulic forces where there is a sufficient supply of coarse sediment. Rivers depend on the movement of coarse sediment between riffles (at high flow) and consistent fine sediment deposition and scouring. In natural environments, rheophilic invertebrates are more abundant and can actively recruit new gravel and maintain a hydraulic maintenance of natural benthic faunas. More prevalent ratios of fine sediments in lowland modified rivers are likely to impair this function (Pulg *et al.*, 2013), with the role of artificial gravel being reduced and instead dominated by fine sediments infiltrating riffles (Kemp *et al.*, 2011). Furthermore, areas of refugia sought out by invertebrates in urban lowlands are likely to be replaced by macrophytes and riparian vegetation, where particle organic matter is not likely to be a limiting factor. Hatched larvae acquire marginal vegetation, disperse into mid-channel benthos and make use of bankside vegetation upon reaching maturity (Harrison and Harris, 2002). In urban streams, poor water quality such as reduced dissolved oxygen and high NH_4^+ is likely to be reducing the functioning of highly oxygenated riffle-pools.

Some studies have highlighted the limitations associated with scale (local vs regional) in determining restoration success (Leps *et al.*, 2016). Not only can catchment-scale characteristics override local restoration efforts, but the potential improvements of restored reaches may be constrained by the regional species pool. This poses important questions surrounding the effectiveness of restoration at larger regional and catchment scales, by recognising functional links between geology, land-use and riparian habitats.

Further considerations should be made towards restoration age and long-term monitoring of restoration activities (Leps *et al.*, 2016).

Observations of the functional links between fish and invertebrate species were made which points to dynamic processes which may be occurring in these streams. Foremost, fish species considered to be less tolerant to pollution (*S. trutta*) were also observed among sensitive invertebrate species (*S. ignita*). The same pattern was noted for highly tolerant invertebrates (Oligochaeta spp. and Erpobdella) and a reduction in fish richness at the Brent, where previous readings elevated NH_4^+ concentrations have been made. This is likely to be a combined factor of water quality, unfavorable substrate composition (as evident at the Hogsmill unrestored site) and in-channel vegetative features. Further differences including food supply, predation and tolerances to pollution which are likely to be contributing to these differences.

At the Wandle, strong predator-prey species interactions between *C. gobio*, *G. pulex* and *S. trutta* were noted. *C. gobio* is extremely abundant in freshwater environments and frequently accounts for a significant proportion of total fish populations in UK rivers (Carter *et al.*, 2004). At the Hogsmill, *C. gobio* accounted for 54% and 95% of individuals caught at the restored and unrestored site, respectively, whereas at the Wandle, this number was 95% and 96%. They are a voracious predator of *G. pulex* (dominant detritivore) and are also preyed upon by *S. trutta* (Woodward *et al.*, 2008). Strong negative density dependencies have been noted between *C. gobio* and *S. trutta*. Woodward *et al.* (2008) also suggested that the negative impact of *C. gobio* on *G. pulex* can suppress leaf litter breakdown rates.

Inverse relationships between fish and invertebrate density and biomass at the Wandle were recorded. At the restored site, *C. gobio* numbers were marginally lower than that of the unrestored site, but overall biomass was greater. The same pattern was observed for

G. pulex, where densities were lower, but biomass values were higher at the restored site when compared with the unrestored site. Possible causes of this may include favourable environments for taxa proliferation and foraging and predation potential. Factors influencing taxa density over short distances (< 200 m) warrant further work, as growth rates and survival of fish and invertebrate species can be influenced by additional factors. Overall, there were a greater number of *S. trutta* and *C. gobio* at the restored Wandle site, reflecting increased productivity. Lower *C. gobio* biomass values at the restored Wandle site when compared to the unrestored site may reflect predation by *S. trutta*.

A significant limitation in this project was the lack of before-after-control-impact (BACI) studies. BACI is an approach used to assess the effectiveness of restoration measures by monitoring pre and post-project conditions (i.e. hydromorphology, biology). Furthermore, untreated (e.g. unrestored) ‘control’ and natural ‘reference’ sites should be incorporated to isolate confounding ‘noise’ and natural variability (Feld *et al.*, 2011). Despite the statistical strength of the BACI approach, few studies have implemented this approach to quantify restoration benefits. To this end, it is difficult to fully measure restoration success.

5.4.5 Implications for Restoration

Studies have documented patterns of degradation and biodiversity loss in urban streams, however few examine how restoration can reverse these impacts (Stranko *et al.*, 2008; Coles *et al.*, 2010). The importance of managing rivers for scale and habitat type has been recognised by several authors (Pretty *et al.*, 2003; Kail and Herring, 2009). Hydrological regimes often operate at scales greater than 2 km, therefore adopting reach-scale approaches may not necessarily elicit ecological recovery (Kail and Herring, 2009). Furthermore, it is often considered to be more effective to restore areas adjacent to reaches where there is greater restoration potential, instead of targeting degraded isolated sections

(Pretty *et al.*, 2003). Pretty *et al.* (2003) suggest that reach-scale restoration in many UK lowland rivers (ranging from 150 to 1000 m), may only be successful if supported by adjacent colonisers that are suitable for the habitat. Alongside this, it is important to consider that favourable riparian vegetation or in-stream structures may only attract fish rather than improve overall productivity (individual size and growth) (Giakoumis and Vouvoulis, 2019). This statement is likely applied to reaches within this study, which require further work to understand community structure over greater spatial scales.

The results from this study show that restoration techniques have the potential to result in reach-scale ecological improvements. Positive reach-scale responses at the Wandle were observed, where fish and invertebrate numbers are likely to have increased due to habitat features, for example macrophytes and riffle-pools providing favourable backwater environments (Miller and Kochel, 2010). Based on these observations, future work could seek to quantify the importance of these features. It is also important to consider that fine sediments are sinks for pollutants and provide hotspots for storage of N (Haase *et al.*, 2013; Chapter 3) which may limit the ability of certain species to return to restored sites despite improved hydromorphology. It is often difficult to manage restored urban lowlands where flood control and land drainage are prioritised over re-instating natural processes. Nevertheless, features along the upstream river, such as macrophytes and riparian vegetation have the potential to remedy some of these effects. Although the method used to examine fish populations was robust, a limitation was that samples were taken during the Autumn so only provided a snapshot of the community composition. This also limited the quantification and assessment of juvenile fish.

Overall mitigation projects are dominated by scientifically weak approaches that lack specific objectives, post-monitoring examinations (both essential for understanding longevity effects) and landscape processes. Many also report from a reach-scale (< 1 km)

which does not consider spatial variability. Lowland urban rivers should seek to utilise natural fluvial dynamics with minimal engineering (Pretty *et al.*, 2003; Harrison *et al.*, 2004), including efforts to make use of lateral space by restructuring floodplains and encouraging emergent macrophytes. Some studies have argued that jetting can provide potential for non-salmonid fish at different stages of their life cycle. For example, spawning of cyprinid fish are asynchronous, therefore jetting may coincide with late-spawning species such as *B. barbatula* and *L. leuciscus* (Maitland and Linsell, 2006). As with other studies where there is an abundance of fine sediment, utilisation of flow deflectors could help to prolong jetting. Managing phenology of different fish within the local community may help to achieve optimal benefits.

5.5 Conclusions

Ecological community structure varied across rivers in this study, revealing some differences between restored and unrestored reaches (e.g. richness). Given the degree of variability and lack of baseline data available for these sites, there is a need to reinstate defined goals for ecological restoration which consider structural and functional objectives. In the case of London rivers, downstream urban catchments dominated by physically modified channels and high nutrient inputs are likely to require longer-term, larger spatial management than that of natural upland environments. Creating favourable environments for fish species may not necessarily elicit ecological improvements. Thus, holistic approaches are required to further understand the anthropogenic factors driving these pressures.

Chapter 6: Summary and Future Research

6.1 Summary of Research Findings

Despite a growing number of river restoration projects being adopted worldwide, there is still insufficient evidence to suggest that modifying physical stream habitats can elicit widespread, ecological and biogeochemical ecosystem improvements (Pretty *et al.*, 2003; Harrison *et al.*, 2004). The ‘Field of Dreams’ hypothesis set out by Palmer (1997), describes the assumption that modifying physical habitat structure can foster ecological recovery. Insufficient evidence to support this theory (Lepori *et al.*, 2005; Palmer *et al.*, 2010; Smith and Chadwick, 2014) has driven investigations of stream ecosystem processes and functions (e.g. productivity and disturbance regimes) to understand process-based determinants of restoration success (e.g. Roberts *et al.*, 2007; Mulholland *et al.*, 2008; Tiegs *et al.*, 2013; Smith and Chadwick, 2014).

Unlike experiments which assess structure (geomorphology, fish and macroinvertebrates), process-based studies can be difficult to implement due to their complexity and lack of standardisation. Furthermore, knowledge of ecosystem functions require crossover between a range of academic disciplines. Despite these challenges, it is increasingly important to explore the underlying processes driving the physical, chemical and ecological composition of urban river ecosystems. In addition, holistic approaches are required to inform effective restoration management practices which provide resilience to future environmental challenges.

The overall aim of this thesis was:

To measure the effectiveness of urban river restoration through examination of reach-scale physical, chemical and biological processes which affect biogeochemical process rates and ecological community structure.

This research project utilised a range of methods to examine ecosystem functions (nitrogen processing, decomposition) and structure (fish and macroinvertebrate community structure) across restored and unrestored reaches of the River Thames in London (Figure 6.1). This thesis provides an empirical platform from which to discuss two key areas of research: a) the processes occurring within urban streams and their influence on ecosystem structure and b) the opportunities for function-based restoration. The key findings from this project, including proposed mechanistic processes responsible for observed results, are discussed in this chapter. The individual and combined findings from each of the data chapters are summarized and linked back to the initial research aims. Subsequently, consideration is given to areas of future study including the development of function-based restoration measures.

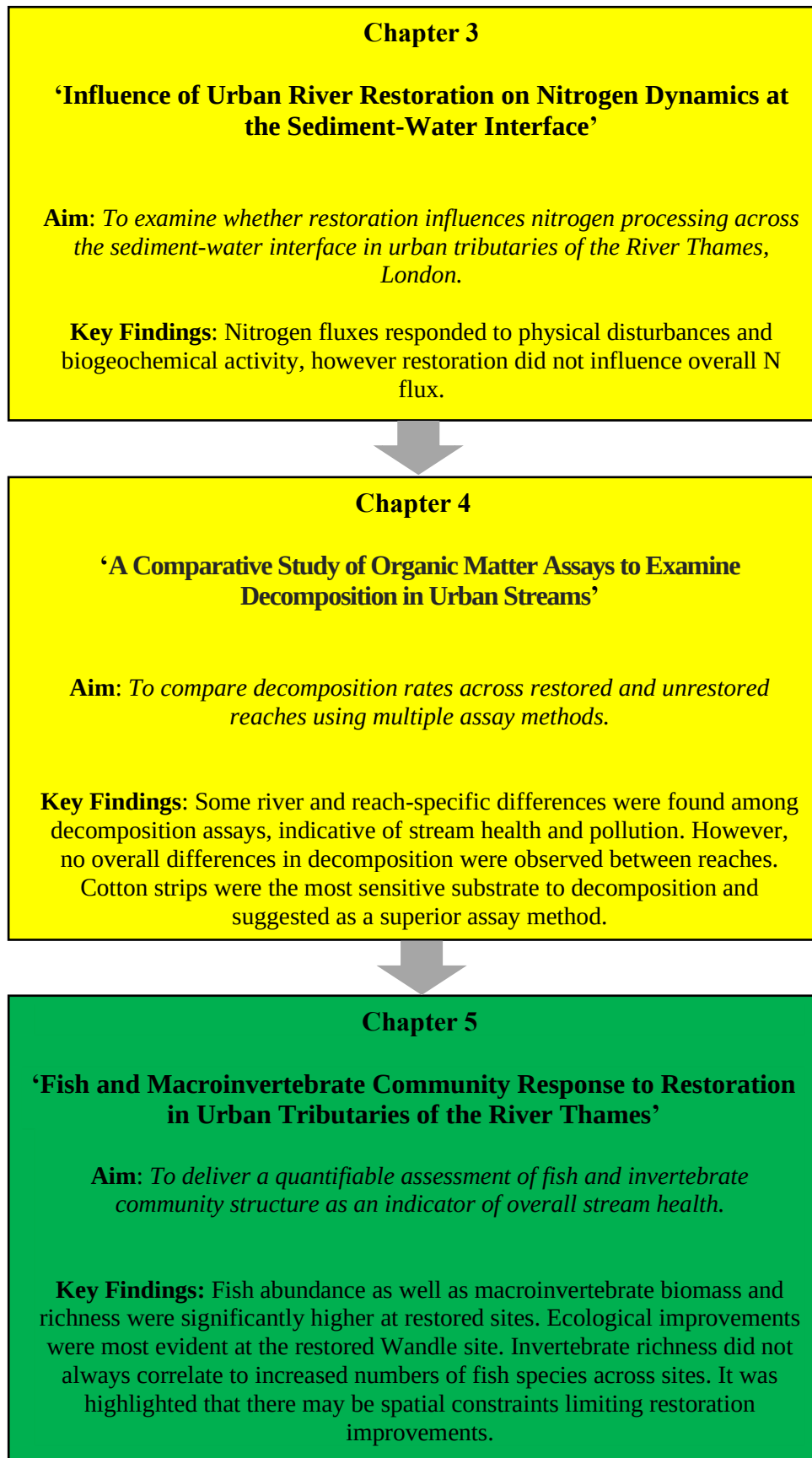


Figure 6.1. Project aims and key findings. Yellow boxes denote empirical data chapters examining functional processes; the green box denotes the data chapter examining ecosystem structure.

Chapter 3: *Influence of Urban River Restoration on Nitrogen Dynamics at the Sediment-Water Interface:*

N pollution in urban freshwater systems is an inherent problem, however few studies have examined the effects of water-sediment interactions on N processing at patch-scales. This chapter utilised a novel flux chamber assay to examine ammonium (NH_4^+) and nitrate (NO_3) processing across the sediment-water interface in urban streams of London. This approach was chosen due to its cost-effectiveness, small-scale application and ease in deployment. The study aimed to understand whether restoration could influence nitrogen processing across the sediment-water interface in urban tributaries of the River Thames, London. It was hypothesised that:

- 1:** Restoration would reduce nitrate and ammonium concentrations and facilitate increased nitrogen uptake at restored sites compared to unrestored sites, owing to the presence of in-stream features driving nitrogen assimilation.
- 2:** Physical disturbances and biogeochemical activity would increase regeneration at unrestored sites due to sediment nutrient saturation and increase uptake at restored sites due to higher sediment sink capacities.
- 3:** Higher concentrations of chlorophyll-*a* and proportions of organic matter would be linked to nitrogen removal due to facilitated assimilative processes, while higher proportions of fine sediment would be linked to nitrogen regeneration due to an increased sediment storage capacity.

0-3-minute flux intervals simulated physical disturbances (i.e. weir removals, flashy flows) characteristic of urban waterways while 3-10 minute fluxes reflected longer-term biogeochemical processes. Previous studies have highlighted the importance of physical

disturbances and biogeochemical activity for driving nutrient transfers within urban streams (Roberts *et al.*, 2007; Mulholland *et al.*, 2008; Hines and Hershey, 2011).

Results revealed that NH_4^+ concentrations were higher at restored sites attributed to the River Brent where concentrations exceeded $700 \mu\text{g l}^{-1}$. This river is subject to substantial nutrient loads and stochastic events have been previously reported (Smith and Chadwick, 2014; Pecorelli, 2017). Average NH_4^+ fluxes did not differ between 0-3 minutes and 3-10 minutes across reaches and rivers in response to disturbance and biogeochemical activity, thus rejecting Hypothesis 1 and 2. However, average NO_3^- fluxes among sites responded significantly between 0-3 minutes. Given that regeneration of NH_4^+ and NO_3^- was not observed from 3-10 minutes, it is suggested that uptake is not occurring and sediments are saturated with N (Walsh *et al.*, 2005; Kaushal and Belt, 2012).

There was no correlation between chlorophyll-*a*, total organic matter, sediment grain size and $\text{NH}_4^+/\text{NO}_3^-$ fluxes, thus rejecting Hypothesis 3. Variations in overall nitrogen (N) fluxes were attributed to N-specific sediment storage capacity, biogeochemical transformations and potential legacy effects associated with urban pollution (Walsh *et al.*, 2005; Kaushal and Belt, 2012). A greater understanding of mechanisms driving the dynamic transfer, processing and removal of NH_4^+ and NO_3^- in urban river systems is required to inform future river restoration strategies. Conclusions highlight the need to foster an improved understanding of the mechanisms controlling the inputs, processing and transformations of NH_4^+ and NO_3^- in urban river systems. This is particularly true for the highly urbanised system found in megacities like London which far exceed impervious cover characteristics of other cities.

Chapter 4: *A Comparative Study of Organic Matter Assays to Examine Decomposition in Urban Streams.*

This chapter used multiple decomposition assays (coarse and fine leaf litter, tea bags and cotton strips) to examine functional ecosystem responses to restoration. The aim of this study was to compare decomposition rates across restored and unrestored reaches using multiple assay methods. Previous studies have attributed breakdown to physical abrasion, temperature, invertebrate shredding, microbial activity and water quality (Pascoal *et al.*, 2005; Chadwick *et al.*, 2006; Tank *et al.*, 2010). It was hypothesised that:

- 1:** Decomposition would differ significantly between restored and unrestored sites, owing to processes including physical abrasion, nutrient processing and biological activity.
- 2:** Cellulose cotton strips would provide a preferential tool to measure decomposition over leaf litter or tea bags to their uniformity and ease of application.

Breakdown rates across leaf litter and tea bag assays (ranging from 0.0001 to 0.002 k_{dd}) were consistent with previous studies (Smith and Chadwick, 2014) which found that there were no differences in leaf litter breakdown between restored and unrestored reaches, thus rejecting Hypothesis 1. However, reach-scale analyses revealed reach-scale differences in decomposition rates in fine mesh bags, primarily driven by higher breakdown at the restored Brent site. In this instance, Hypothesis 1 is somewhat accepted. Increased breakdown rates in coarse mesh compared with fine mesh bags may suggest increased physical abrasion and shredder activity (Chadwick *et al.*, 2006; Tank *et al.*, 2010), however these drivers can only be inferred. Conversely, tea bags exhibited breakdown rates which were higher than that of leaf litter in coarse and fine mesh bags, suggesting that tea leaf substrate may provide a preferential food source for invertebrates and microbes.

Rates of decomposition for tea bags were on average twice as high as leaf litter, however no differences were noted between rivers or reaches ($p = 0.59$; $p = 0.67$). Cotton strip breakdown (% loss/degree day) did not vary between rivers or reaches ($p = 0.94$; $p = 0.83$). Cotton strips showed the largest percentage loss compared with other assays used. These findings are in line with previous work that credit the sensitivity of cotton strips in measuring decomposition (Tiegs *et al.*, 2007; Tiegs *et al.*, 2013), and support Hypothesis 2. The limitations of using conventional leaf litter and tea bags assays are also explained. However, differences in breakdown rates highlight the importance of standardisation and consistency among assays so that direct comparisons can be made across different geographical scales. These findings also necessitate future work to understand the contribution of fungi and microbes to decomposition, which may be reflective of both ecological integrity and increased pollution.

Chapter 5: *Fish and Macroinvertebrate Community Responses to Restoration in urban tributaries of the River Thames.*

This chapter examined stream responses of fish and invertebrate community structure to restoration as determinants of environmental health. The main aim was to deliver a quantifiable assessment of fish and invertebrate community structure as an indicator of overall stream health. Previous studies have highlighted the benefits that habitat manipulations can bring for increasing fish and invertebrate abundances (Pretty *et al.*, 2003; Harrison *et al.*, 2004).

It was hypothesised that:

1: Fish abundance, biomass and richness would increase at restored sites due to improvements in hydromorphological conditions fostering ecological integrity.

2: Macroinvertebrate density, biomass and richness would increase at restored sites due to improvements in water and habitat quality.

3: Physical habitat heterogeneity would increase at restored sites due to improvements in hydrological heterogeneity and habitat diversity.

Fish abundance and invertebrate biomass were higher at restored sites, indicative of improvements in habitat creation suitable for fostering ecological recovery (supporting Hypothesis 1). A total of nine fish species (comprising 1541 individuals); 955 at restored reaches and 586 at unrestored reaches, were recorded. The highest estimates were recorded at the Wandle attributed to the dominance of *C. gobio*. The most significant improvements in biomass and richness were observed at the Wandle, where *S. trutta* was recorded alongside sensitive species including *S. ignita* and Limnephilidae spp. Stocking has significantly influenced *S. trutta* abundance along this reach by subsidising species numbers and acting as a mechanism for dispersal (Jenkins *et al.*, 1999). Some studies have highlighted the importance of gravel bed spawning grounds for the survival of salmonids in chalk streams, further providing high-quality spawning grounds (Pulg *et al.*, 2013).

Interactions were observed between *C. gobio* and *G. pulex*, highlighting important predator-prey connections indicative of improved ecosystem functioning (Woodward *et al.*, 2012). The Pool and Brent were the only rivers where biomass values were greater at unrestored reaches, owing to increased numbers of *Leuciscus leuciscus* (999 g) and *A. anguilla* (153 g), respectively. The Pool was also the only site where perch *Perca fluviatilis* and roach *Rutilus rutilus* were recorded. Preferential habitats were also highlighted for certain species, for example *R. rutilus* and *P. fluviatilis*, which favour deep, slower moving waters. The Brent restored site was in a worse ecological condition than the paired unrestored site, with significantly more Oligochaeta spp. and reduced fish

abundance and biomass. Poor water quality previously recorded at this site is likely to be a contributing factor to this (Chapter 3).

Invertebrate communities increased in biomass and richness at restored sites compared to unrestored sites, but not in density, somewhat supporting Hypothesis 2. SIMPER analysis revealed that *G.pulex* and *A.aquaticus* were the dominant species driving differences among sites. Physical habitat characteristics did not significantly influence fish and invertebrate abundance, density and biomass, except for the Hogsmill and Brent (somewhat supporting Hypothesis 3). Physical habitat assessments throughout this project were variable and highlight the stochastic nature of urban streams.

6.2 The Utility of Restoration for Improving Ecosystem Function

The question of whether restoration is ‘an active cure or just a placebo’ (Feld *et al.*, 2011) is widely debated for several reasons. Not least, there is a deficiency of robust data which examines pre-and post-monitoring conditions (e.g. hydromorphological conditions, biological community structure) related to restoration activity. A scientifically sound approach to assess restoration success is through the before-after-control-impact (BACI) approach. This examines pre (before) and post (after) monitoring conditions, in addition to untreated ‘control’ sites and natural ‘reference’ sites (Feld *et al.*, 2011). Furthermore, it also accounts for natural variability (e.g. seasonality) in streams. Worldwide, there are few BACI studies associated with river restoration projects. To this end, it is difficult to accurately quantify the degree of stream improvements. Given that many sites examined in this study had minimal background environmental quality data, it was not possible to conduct a BACI assessment. Instead, paired restored and unrestored reaches were selected in close spatial proximity to one another to allow for reach-scale comparisons. Kemp (2011) emphasised the importance of understanding and managing habitats to facilitate different functions for species that operate at a range of scales (meso to macro).

An example of this is in circumstances where fish require gravel beds for spawning, but connectivity for migration at a catchment-scale.

Restoration efforts at the selected sites did not influence overall nitrate or ammonium processing (Chapter 3) (Figure 6.2). Several studies have shown that physical modifications can encourage temporary or permanent N removal (Newcomer Johnson *et al.*, 2016) and this study sought to explore this assumption (Craig *et al.*, 2008). The failure of these reaches to remove N may be down to several factors, including the scale of N processing which occurs over much larger catchment-scales, saturation levels and the failure of current structures to facilitate N removal. Filoso and Palmer (2011) found that the majority of N removal in watersheds occurred before entering streams, which suggests that more is needed to understand catchment-scale processes controlling N processing. It is highly likely that high N loading in these streams, legacy effects and failure of in-stream structures (riparian vegetation) to remove nutrients, may explain the failure of these reaches to remove N (Burt *et al.*, 2010; Figure 6.2). Howden *et al.* (2010) has suggested that groundwater flow times may also been a contributor to increased N along the River Thames in recent decades. However, Worrall *et al.* (2009) has outlined the importance of long-term nitrogen accumulation in soils and leaching losses of dissolved organic nitrogen from soils driven by climate.

Another measure of ecosystem function, organic matter decomposition, revealed equivocal differences in breakdown between restored and unrestored sites for all assays. An exception to this was elevated breakdown rates observed in fine litter bags at the Brent restored site ($p = 0.009$). The mechanisms driving this change can be assumed from previous studies which have attributed increased decomposition to microbial activity and physical abrasion (Meyer *et al.*, 2005; Chadwick *et al.*, 2006) (Figure 6.2). Given that the Brent was the most polluted river, it is possible that nutrient pollution may have

contributed to breakdown at this site. However, a major challenge in urban river restoration studies is untangling different “compensatory effects”. For instance, increased breakdown rates can be a result of shredder activity (characteristic of good quality), in addition to nutrient pollution. Ecological structural improvements were evident across all restored reaches, with notable increases in fish abundance, invertebrate richness and sensitive species. Elevated ammonium levels at the Brent are likely to have contributed to increased rates of cotton strip decay. This coincided with a lower abundance and richness of fish and invertebrate species. One potential reason for the failure of restoration projects is the spatial scale over which most works take place (Feld *et al.*, 2011). Most reach-scale restoration projects occur at reach-scales, however functional processes (e.g. nutrient processing) and structural indicators (e.g. fish and macroinvertebrates) respond to these factors over much larger spatial scales.

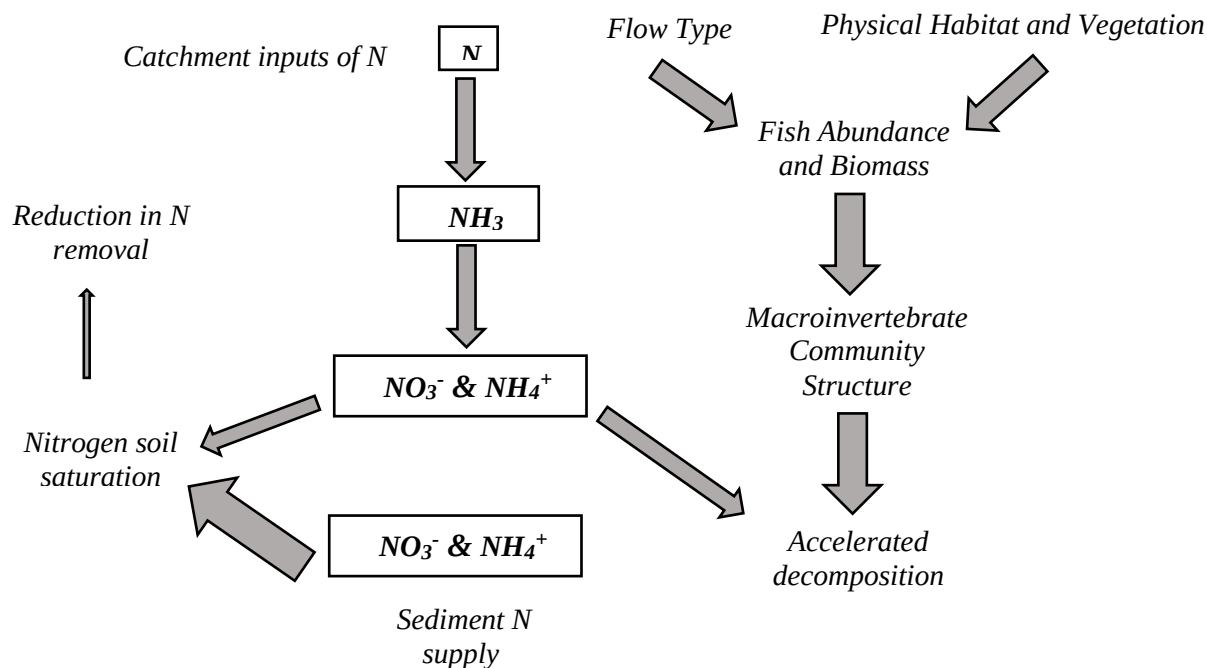


Figure 6.2. Schematic illustrating proposed processes eliciting observed ecosystem responses in this study. Diagram shows minimal reductions in nitrogen removal and high sediment N supply to overlying waters. Accelerated decomposition rates at the River Brent are linked to high NH_4^+ concentrations, although the role of nutrient processing on breakdown cannot be assumed for all sites. Physical habitat structure, vegetation and flow are likely controls on fish and macroinvertebrate community structure, however more research is needed to understand these processes.

6.3 Directions of Future Study

The importance of examining ecosystem function in river restoration projects is often overlooked in favour of examining structural indicators. This is reflected in the application of ‘restoration ecology’ – the idea that habitat heterogeneity will foster improvements in biodiversity. The choice to take this approach is understandable given that physical habitat assessments involve more tractable methods to measures ‘success’ than assessing processes driving these changes such as productivity and disturbance regimes (Palmer *et al.*, 2010). However, few studies support the view that habitat change alone can drive habitat improvements. Feld *et al.* (2011) suggested that research should instead focus on understanding processes linked to restoration success, for example erosion and deposition which influence flow regimes (e.g. pool-riffle sequences). It is important to recognise that restoration success may be limited by physical constraints, for example agricultural regimes which may overshadow reach-scale restoration efforts. The disparity between geomorphological and ecological restoration should be considered in addition to ecological mesohabitats (e.g. fine sediment zones) which are functionally important for species such as juvenile lampreys, which may score poorly in geomorphological assessments (Feld *et al.*, 2011).

Patch-scale approaches undertaken in this work necessitate future work to understand nitrogen activity at catchment-scales. Research related to nitrogen processing has tended to concentrate on the removal of anthropogenic nitrate via ‘denitrification’ at patch (e.g. discrete samples of gravel or piezometers) and catchment-wide scales, rather than considering the whole nitrogen cycle (Burgin and Hamilton, 2007). Future work should aim to understand the driving factors influencing N uptake, including exploring seasonal differences. Future research should incorporate tracer studies to determine the source, saturation concentrations and fate of N. Additional restoration of urban systems would benefit from upscaling studies by looking at different patch types, especially larger calibre

sediment types typically associated with restoration practices (e.g. additions of gravels). Future studies should seek to examine the role of substrate and macrophytes in delivering ecosystem improvements (Kemp *et al.*, 2011) and examine impacts on fish and invertebrate assemblages by understanding habitats and diet (Woodward *et al.*, 2012). Additionally, future restoration projects focused on increasing fish spawning would benefit from increased monitoring of population level characteristics, for example reproduction, history and growth.

6.4 Concluding Remarks

The holistic approach taken in this PhD allowed for the assessment of a wide range of structural and functional parameters to examine ecosystem functioning across urban tributaries of the River Thames. Given that there is little or no baseline monitoring data for these reaches, combined with the fact that that restoration at the sites occurred over different timescales, comparable assessments of river restoration projects can provide a relative baseline for informing future work. While it is difficult to determine the scale and rate of improvement given the lack of information pre-restoration, this study has revealed marked potential for ecological improvements at restored reaches. These results highlight the need to decipher drivers of ecological recovery through functional stream assessments focused on stream processes, including primary production.

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Appendix 1: POSTnote on Environmental Principles

A three-month fellowship was undertaken during this PhD at the Parliamentary Office of Science and Technology (POST): <https://www.parliament.uk/mps-lords-and-offices/offices/bicameral/post/>. POST is Parliament's in-house source of scientific advice and offers advice on key research evidence relating to public policy issues. During this fellowship, I produced a POSTnote in Environmental Principles, summarising the key principles (i.e. the precautionary principle, prevention principle, rectification at source principle and polluter pays principle) and considering the potential opportunities and challenges surrounding their implementation post-Brexit. This note was informed by several key experts, including academics, NGOs and industry partners.



EU Environmental Principles



Environmental principles inform legal and political frameworks that aim to minimise the ill-effects of human activity on the environment. In the EU (Withdrawal Act) 2018, the UK has committed to incorporating a set of environmental principles into UK legislation. This POSTnote summarises these principles and considers potential opportunities and challenges surrounding their implementation post-Brexit.

Background

Environmental principles inform legal frameworks that relate to environmental protection or sustainable development.¹ One group of environmental principles has been used in EU policy-making since the 1970s, and a wider set of principles were agreed globally at the 1992 Rio Declaration on Environment and Development.^{2,3} They are also nested in a whole array of broader principles (including proportionality and subsidiarity outlined in the Treaty on the Functioning of the European Union (TFEU) articles 3, 5, 9–12). These broader principles apply to the environmental policy realm but are not specifically 'environmental'.

Article 191(2) of TFEU sets out four main environmental principles that must guide policy within the scope of EU law (CBP8132).⁴ These have been influential in formulating a range of EU directives and regulatory actions, and will constitute the main focus of this note. The principles listed are:

- The precautionary principle, which allows regulatory action to be taken even if a risk has not been established with full certainty.⁵
- The prevention principle, which aims to prevent environmental damage; such as to protected species or to natural habitats, water and soil; rather than to react to it.⁶

Overview

- EU environmental law and policy is based on four core environmental principles contained in Article 191(2) of the Treaty on the Functioning of the European Union (TFEU): the precautionary, prevention, rectification at source, and polluter pays principles.
- The principles do not create direct legal rights but have been used by the courts to interpret and apply EU environmental law.
- The precautionary principle is applied to manage risk in cases of scientific uncertainty.
- The polluter pays principle is used to allocate responsibility for pollution costs, although attributing these costs can be complex.
- There is ongoing discussion around future implementation and adoption of the principles in the UK post-Brexit.

- The rectification at source principle, which seeks to prevent pollution at its source rather than remedy its effects.^{7,8}
- The polluter pays principle, which requires polluters to bear the financial cost of their actions.⁸

European Union (Withdrawal) Act 2018

In Section 16(2) of the EU (Withdrawal) Act 2018 along with the four core principles, a further five principles are outlined.⁹ Two of these are contained in Article 11 of TFEU:

- The integration principle, which requires environmental protection measures to be integrated into all EU policies and activities, with a particular view to promote sustainable development.^{9,6,10}
- Sustainable development, which is not defined in TFEU, but widely defined internationally as development that aims to "meet the needs of the present while not compromising the ability of future generations to meet their own needs" (POSTnote 408).^{11,12}

The other three principles are qualitatively different from the four core principles, outlining not general ideas of policymaking, but legal rights for individuals around which there is extensive jurisprudence. These ensure that the UK continues to live up to its Aarhus commitments without the implementation structure of EU law:

- Public access to environmental information, which ensures access to written, visual, aural or database

information held by public authorities concerning the environment.¹³

- **Public participation in environmental decision-making**, which relates to the right of public participation in public decision-making relating to environmental matters.
- **Access to justice in relation to environmental matters**, which enables individuals and their associations to exercise rights granted to them under the Aarhus Convention.¹⁴

The Agreement on the withdrawal of the United Kingdom of Great Britain and Northern Ireland from the European Union and the European Atomic Energy Community requires the UK to respect the four main principles outlined in TFEU in environmental legislation.¹⁵ This POSTnote provides an overview of these, examining their merits, challenges and options for implementation in the UK post-Brexit.

The Precautionary Principle

The precautionary principle was defined in the 1992 Rio declaration as “where there are threats of serious or irreversible damage, lack of full scientific certainty shall not be used as a reason for postponing cost-effective measures to prevent environment degradation.”³ The principle has been applied globally to guide policy on issues such as food safety, air quality, acid rain, climate change, North Sea pollution and most frequently to the restriction of chemicals.^{16–20} EU applications of the principle include:

- The Environmental Quality Standards Directive 2013/39 EU,²¹ which sets environmental quality standards for priority substances on a precautionary basis. For example, persistent organic pollutants, potentially toxic metals (POSTnote 579) such as cadmium,²² and polycyclic aromatic hydrocarbons²³ are subject to standards set to achieve ‘good’ water quality as required by the Water Framework Directive (WFD).²⁴
- Article 6(3) of the Habitats Directive (92/43/EEC), which requires impact assessments to be carried out where a plan or project is likely to have a significant effect on the integrity of a designated habitat site.²⁵ The UK Government was ruled against by the European Court of Justice for not complying with the Directive when allowing water abstraction in wetland sites designated under the Directive.²⁵
- The deliberate release of Genetically Modified Organisms (GMO Directive) 2001/18,²⁶ including requirements for field testing in the research and development stage assessing how their use might affect ecosystems,²⁷ which is discussed below.

The precautionary principle has been advocated as a benchmark for regulating risk across different legal cultures.²⁸ However, translating scientific uncertainty into legal obligations requires a careful and nuanced understanding of both.^{29,30} Decision-makers need to have rational and accountable decision-making processes for addressing risks. This involves applying tests of administrative review to understand the legality of decision-making. A report for the European Commission (EC) Directorate General Environment highlighted that actions guided by the precautionary principle should be based on

costs, benefits and best available scientific evidence. However, it also stated that an advantage of the approach is the overt recognition of uncertainties and the negotiated nature of decision-making.⁵ The precautionary principle is not intended to eliminate risk, but ensure that the benefits of a decision outweigh the risks. For example, chlorine is often used for its antimicrobial properties to protect human health, but poses a health risk in high concentrations.³¹ A precautionary approach is applied when there is a risk of harm, and scientific knowledge is not sufficient to reduce this risk. If harm can be demonstrated, a preventative approach is taken (below and Box 1).¹⁹

The Precautionary Principle and GM Crops

In 1998, the EC started a negotiation on new genetically modified (GM) legislation,³² halting the application process for approving GM Crops. In 2003, the US started a World Trade Organization (WTO) case against the EU on the grounds that the approval of biotech products was so slow that it amounted to a moratorium. In this case, the EU invoked the precautionary principle.³³ It contended in particular that the 2003 Cartagena Protocol on Biosafety – to which the EU, but not the United States, is a party – permits states to adopt a precautionary approach toward products created with new technologies.³⁴ Although the dispute settlement panel found in favour of the US, it did not state any opinions on the legality of future EU measures or whether the precautionary principles is part of international law binding on all states.³⁴

Box 1: Neonicotinoids and Bees

Neonicotinoids are a class of pesticides which have been associated with declines in bee populations.³⁵ In December 2013, acting on advice provided by the European Food Safety Authority, the EC triggered an open-ended restriction on three neonicotinoids linked to impacts on bees used for seed treatments, soil applications and foliar sprays (SN06656).^{36–38} These measures were based on concerns that they could have harmful, but not lethal, effects on bees (SN06656). However, at the Standing Committee for Food Chain and Animal Health, and a submission to the Appeal Committee, there was no qualified majority support for the European Commission’s proposal to restrict neonicotinoids. As a result, the Commission took the decision on the restrictions itself.

In 2018, an EU-wide restriction on outdoor neonicotinoid use will come into place.^{38,39} While the precautionary principle is not always about imposing bans, the prevention principle may require such restrictions. The restrictions on their use followed scientific debate, which concluded that there was a greater risk of harm to bees than had been previously understood.^{40,41} A report by the European Food Safety Authority highlighted that bees are exposed to harmful levels of neonicotinoids in treated pollen, nectar and drifting dust.⁴² Some studies have outlined the need for landscape-scale experiments in real conditions to be undertaken to determine the effects of neonicotinoids on bees at different spatial and temporal scales.^{40, 43, 44} Further studies have highlighted the need for further measures to avoid the use of other, more environmentally harmful chemicals in response to the ban (SN06656). As one of the authors of an academic paper, the Defra Chief Scientific Advisor has suggested monitoring of the use and effects of pesticides are required at the landscape level to inform risk-based pesticide use, which makes the trade-offs between the environmental effects and food production more explicit.⁴⁵

The links between the low levels of GM crops grown in EU countries and the precautionary principle are not straightforward.⁴⁶ In the UK, public opinion opposing GM increased alongside challenges surrounding food safety (arising from issues such as the transmission of bovine spongiform encephalitis to humans^{47,48} and *Listeria* food poisoning from cheese).⁴⁹ This was coupled with additional public concerns that biotechnology companies were exercising too much control over consumers' choices.⁵⁰ Perceptions of risk can be shaped by the nature of the risk and the context within which it occurs (POSTnote 564).⁵¹ This will influence regulatory approaches as well as legislative principles. In 2015, the House of Commons Science and Technology Committee recommended that when decisions are made about emerging issues on science and technology, a broad range of social and ethical factors should be taken into consideration. They also recommended a permanent 'Citizens Council' to provide advice on potential social and ethical impacts to the Advisory Committee of Releases to the Environment (whose remit includes Genetically Modified Organisms).⁵²

The Prevention Principle

The prevention principle is intended to prevent, rather than react to, environmental harm from unregulated action.⁶ Unlike the precautionary principle, it is applied in law and policy when the risk of harm to the environment is clear.⁵³ However, the precautionary and prevention principles have been closely linked to one another;⁵³ for example, in the case of ozone-depleting chemicals.¹⁹ In the 1970s, there was general consensus (but no proof) that chlorofluorocarbons could destroy the ozone layer. Thus, their use was cautioned (precautionary).¹⁹ By the late 1980s, scientific evidence emerged that depletion of the stratospheric ozone layer increased ultraviolet radiation exposure, exacerbating the risk of skin cancers and cataracts in humans and animals. This prompted a preventive approach, requiring the phase-out of chlorofluorocarbons in middle income countries within 10 years and low income countries within 15 years.⁵⁴ Until the agreement of the Montreal Protocol in 1987, there was uncertainty as to which principle was being relied upon, but there was scientific consensus on the risk of harm by the time the Protocol came into force in 1989.¹⁹

The prevention principle was one of 11 objectives and principles listed in the First EU Environmental Action Programme in 1973.⁵⁵ In 1983, in the EC's Third Environmental Action Plan it was applied to waste policy (e.g. incineration, landfill and wastewater).^{6,56} The Basel Convention on the Control of Transboundary Movement of Hazardous Wastes and their Disposal (1989) was the most comprehensive global treaty on waste, forming a basis for several applications of the prevention principle in EU law.⁵⁷

Rectification at Source Principle

The rectification at source principle guides the regulation of pollution from its source rather than in the wider environment.⁵⁸ It is not applied as an absolute rule, but as an overriding guide to policy; for instance, encouraging the

development of environmentally friendly technologies and products to reduce pollution at the earliest stage.⁵⁹ It has also been used to inform legal interpretation of some EU environmental regulation.^{1,60}

The Polluter Pays Principle

The polluter pays principle (PPP) asserts that those who produce pollution should bear the environmental and social costs of their actions. The PPP was first recognised by an international body in 1972, the Organisation for Economic Co-operation and Development.⁶¹ Prior to this, air and water resources were used as 'sinks' for pollution, with damage to human health and property being paid for by society rather than by the polluter.⁶¹ Disproportionate social and private costs of pollution were being 'externalised' from the polluter to wider society. The PPP aims to overcome these defaults by requiring polluters to internalise the cost of potential pollution in the production process (built-in costs), rather than allowing society to incur costs in the aftermath.⁶²

Designating Costs

The PPP has been utilised as an economic tool for managing different types of environmental pollution through embodiment in legislation including: The Waste Framework Directive,⁶³ Landfill Directive⁶⁴ and Water Framework Directive.⁶⁵ However, it is often difficult to define the polluter, the source of pollution,⁶⁶ and the associated liability cost over time. Where pollution derives from multiple sources, it should be designated to all polluters.

For example, the Nitrates Directive forms an integral part of the WFD and is one of the key instruments in the protection of waters from agricultural sources. Nitrate pollution may derive from multiple sources in river catchments (Box 2). Stakeholders have also been penalised under the WFD for the failure to meet water quality standards.⁶⁷ However, even when the pollution can be attributed to only one stakeholder, the PPP does not necessarily deter pollution by those who can afford to pay the cost.⁶⁸

The Environmental Principles after Brexit

The European (Withdrawal) Act 2018 outlined a set of 9 environmental principles to provide a basis for future environmental policy action in England.⁹ Provisions in section 16 of the Act 2018 require the Government to publish the draft Environmental Principles and Governance

Box 2: Designation of Nitrate Vulnerable Zones

In 1997, two farmers challenged the UK Environment Secretary of State on the implementation of the Nitrates Directive across three UK rivers (Waveney, Blackwater and Chelmer).^{69,70} They argued that in designated zones where nitrate concentrations had exceeded the Directive limit of 50 mg/l, non-agricultural sources of nitrates were a major contributory factor (such sources include transport, power station and domestic heating⁷¹). On reference to the European Courts of Justice, the High Court of Justice held that "... as regards the polluter pays principle, the directive does not mean that farmers must take on burdens for the elimination of pollution to which they have not contributed ... viewed in this light, the polluter pays principle reflects the principle of proportionality".⁷²

Bill within six months (CBP8132), which must include a set of environmental principles and a policy statement relating to their application and interpretation. Several groups, including the British Academy have outlined the importance of environmental governance arrangements for the application of the principles in their response to Defra's consultation on the principles, which preceded the Act.⁷³

Challenges and Opportunities

There are a number of challenges and closely related environmental governance issues associated with the implementation of the environmental principles in the UK. These include:

- The proposed independent watchdog that might advise or potentially sanction the UK Government in relation to applying the principles.^{74,75} The European Commission enforces EU legal obligations, which may be interpreted or reviewed in accordance with the TFEU environmental principles. The General Council of the Bar of England and Wales (the Bar Council) recently highlighted that the watchdog (as proposed in Defra consultation) would lack equivalent powers to those currently available to the EC and other related legal bodies.⁷⁶
- Developing consistent UK-wide principles and a mechanism for enforcement on all public bodies in relation to applying the principles (see below).⁷⁷ At present the principles will only apply to England, and there are concerns that the UK Government could make decisions that do not fit with principles set by the National Assembly of Wales and those in the Scottish Continuity Bill.^{78,79} This could result in different emerging sets of standards running in parallel in Wales or Scotland and being applied to devolved matters.
- Some organisations have raised concerns that trade with non-EU countries may be prioritised over strengthened environmental standards,⁸⁰ which could hamper adherence to the principles in future.

Application of the Principles by Public Bodies

The House of Commons Environmental Audit Committee Report on the Government's 25-Year Plan for the Environment highlighted the importance of the principles being applied to all public bodies, including non-departmental public bodies and local authorities, and not just central Government.⁷⁷ The view that principles should be applied across all government departments and local government was also outlined by stakeholders in evidence to a National Assembly for Wales Climate Change, Environment and Rural Affairs Committee report.⁷⁸ Given concerns over weak legislative language such as the Government 'having regard to' the principles and their application,⁷⁷ establishing legislative language requiring all public bodies to comply with the principles would represent a greater commitment to the environment.^{77,81}

Potential Additional Environmental Principles

As well as retaining EU principles in UK law, several organisations have highlighted the options for incorporating

additional environmental principles.^{77,76,82} These include the principle of non-regression, which articulates that there should not be any lowering of environmental standards, ambition or protection.⁸³ An example of its application would be continued commitment to the mitigation of climate change at both a national and international level.⁸⁴ This would ensure that environmental standards are not rolled back over time; for instance, when new trade agreements are being negotiated.

Witnesses to the EAC inquiry also referred to a 'high level of environmental protection', outlined in Article 191(2) of the TFEU.⁷⁷ The Government has also announced in the 25-Year plan to incorporate 'net gain' into development, meaning that development projects must improve environmental quality or equal any damage caused.⁷⁷ Other academics have described developing an 'environmental advancement' principle aimed at pursuing increasingly high environmental standards.⁸⁴ Other suggested principles include using the 'best available scientific evidence' to inform policy action and the principle of management at an appropriate physical scale, such as the river catchment.^{77,74,85}

Other commentators have highlighted the issue of adopting other non-environmental EU principles, such as the 'innovation principle', which the EU Political Strategy Centre described as a 'counter principle' to the precautionary principle.⁸⁶ It is defined as the objective of a "highly competitive social market economy, aiming at full employment and social progress, and a high level of protection and improvement of the quality of the environment" (Article 3(3) TEU). Its primary aim is to encourage sustainability and employment, while establishing higher productivity and competitiveness, including social and environmental benefits.⁸⁶

Application and Interpretation of the Principles

Environmental principles have the potential to shape decision-making by the UK government and public bodies. The Government's forthcoming statutory statement of policy on the environmental principles application and interpretation will largely determine how they are implemented. The Environmental Audit Committee recommended this policy statement will require robust scrutiny and consultation to ensure that environmental standards are maintained or improved.⁷⁷ In the 25-Year Plan for the Environment, the UK Government stated its ambition to leave the environment in a better state than that it was inherited.⁸⁷ However, realising this ambition may depend on the effective implementation of the environmental principles in policy frameworks, the sanctioning regime adopted for not doing so, and regulatory frameworks being afforded scope to incorporate additional principles in future.