Impact of Nutrient Enrichment and Hydromorphological Modification on Riverine Biodiversity in Northern Ireland

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Contents

Abbreviations

ACP	Agricultural Catchments Programme
AFBI	Agri-food and Bioscience Institute
AICBRN	All-Ireland Climate and Biodiversity Network
AMBER	Adaptive Management of Barrier in European Rivers
ASSI	Areas of Special Scientific Interest
BAP	Biodiversity Action Plan
BD	Birds Directive
BOD	Biochemical Oxygen Demand
CAP	Common Agricultural Policy
CBD	Convention on Biological Diversity
CEE	Collaboration for Environmental Evidence
CSO	Combined sewer overflows
DPSIR	driver–pressure–state–impact–response
DTC	Demonstration Test Catchments
DWWTS	Domestic Wastewater Treatment Systems
EBVs	Essential Biodiversity Variables
EC	Emerging containments
E-flow	Environment flow
EP	Emerging or re-emerging pathogens
EPA	Environment Protection Agency
EU	European Union
GEO BON	Group on Earth Observations, Biodiversity Observation Network
GES	Good Ecological Status
HaD	Habitats Directive
HMWB	Hard Modified Water Bodies
IFI	Inland Fisheries Ireland
IPBES	Intergovernmental Platform for Biodiversity and Ecosystem Services
IUCN	International Union for Conservation of Nature
IWRM	Integrated Water Resource Management
LAM	Load Apportionment Model

MS	Member States
N	Nitrogen
NBP	National Barriers Programme
NI	Northern Ireland
NiD	Nitrates Directive
NIEA	Northern Ireland Environment Agency
NIW	Northern Ireland Water
OEP	Office for Environmental Protection
OSWWTS	On-site wastewater treatment systems
P	Phosphorus
PP	Particulate phosphorus
PTP	Package treatment plants
RBMP	River Basin Management Plan
RCC	River Continuum Concept
RePhoKUs	Re-focus Phosphorus use in the UK Food System
ROI	Republic of Ireland
SAC	Special Areas of Conservation
SDG	Sustainable Development Goals
SFA	Substance Flow Analysis
SLAM	Source Load Apportionment Model
SPA	Special Protection Areas
SSN	Small Stream Network
SSSI	Sites of Special Scientific Interest
STS	Septic Tank Systems
UK	United Kingdom
UKTAG	UK Technical Advisory Group
UN	United Nations
URF	Urban Runoff
USF	Urban Stormwater Runoff
UWWTD	Urban Wastewater Treatment Directive
WFD	Water Framework Directive
WWTP	Wastewater Treatment Plants
WWTW	Wastewater Treatment Works

Executive summary

The purpose of the following review is to help the Office for Environmental Protection (OEP) obtain a comprehensive understanding of the impact of hydromorphological changes and nutrient enrichment on river biodiversity in Northern Ireland (NI). It will further identify key evidence gaps within this area and provide recommendations for future focus and action.

The Convention on Biological Diversity (CBD, 1992) defined biodiversity as: '*The variability among living organisms from all sources, including inter-alia, terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are part; this includes diversity within species, between species and of ecosystems*". As such, biodiversity refers to the wealth of all life on earth and the links between organisms, including their communities and the habitats which support them. This review focuses on biodiversity in the context of the riverine ecosystem in NI, studying the links between organisms and their habitats within this sphere, along with the key pressures they face. These have been identified as being primarily nutrient enrichment from Phosphorus (P) and Nitrogen (N), along with hydromorphological modification (i.e. the effects of physical barriers and arterial drainage).

As such, the review presents the key drivers relating to biodiversity impairment and loss in river systems across NI, alongside their origins and evolution. The current state of the river environment is also presented, as documented through national monitoring programmes. Recommendations are subsequently made with regards to addressing these issues in order to protect and restore biodiversity in NI – and why that is so important – with gaps in evidence identified. This review should therefore be viewed as a contribution to the wider work associated with riverine habitats in NI.

With freshwater ecosystems amongst the most biodiverse – supporting more than 10% of all known species – they provide multiple benefits to humans, as well as to a wealth of plant and animal species. From being vital sources of drinking water to assisting with agriculture, irrigation, waste disposal and more, rivers add value to our lives and play a key role in society, both economically and socially. As a result, human settlements have historically been established close to rivers, which has subsequently put pressure on these ecosystems, with their resources exploited. This has created pollution 'hotspots' in the surrounding catchments, primarily due to urbanisation and agricultural land use, which has adversely affected the natural character of rivers and resulted in the loss of habitat, biodiversity and wider ecosystem services. These issues have been further compounded by climate and other human-dependant factors such as invasive species and both priority and emerging pollutants, like pesticides and pharmaceuticals.

With the main sources of nutrients within catchments attributed to wastewater, along with diffuse losses from agriculture – including small point sources like farmyards – this review therefore looks at these areas in detail. It presents key information on their impacts upon riverine biodiversity and what is required to manage these – for example, better nutrient source apportionment at catchment scale. Meanwhile, hydromorphological pressures such as flow modification, channelisation, connectivity and physical barriers along the river network are also addressed, along with poor riparian management and land drainage.

With insufficient consideration given to riverine biodiversity to date, this has impeded investment in appropriate policy and management measures, as exemplified through wastewater management in NI. This review therefore highlights the fact that the protection and restoration of ecological integrity and biodiversity requires an acknowledgement of the holistic nature of rivers, integrating sound scientific principles with management perspectives and policy ambitions which promote sustainable environmental heterogeneity.

Ultimately, rivers are diverse ecosystems which offer an abundance of opportunities to society, as well as to the flora and fauna which live within the riverine environment. They have played a vital role in our culture throughout history and, in order to safeguard this valuable resource and ensure its future success, protecting and sustaining the biodiversity of rivers is of the utmost priority. This review therefore seeks to outline the key factors currently affecting riverine biodiversity – to enhance understanding within this area – and to act as a reference with regards to working towards positive solutions to protect it.

Chapter 1: Introduction

Freshwater ecosystems relative to their area are among the most biodiverse ecosystems, supporting over 10% of all known species (Dudgeon et al. 2006; Vörösmarty et al. 2010). Rivers are unique ecosystems characterised by running water and are amongst the oldest and complex components of the freshwater ecosystem, supporting diverse habitats, aquatic species and wildlife (Poff, 1997). They subsequently provide the interface between water bodies and their catchment areas, including riparian zones and floodplains, with their spatial arrangement into dendritic arborescence – together with the inherent downstream-directed water flow – making them unique (Alexander et al., 2007). These two characteristics affect not only the chemical composition and physical architecture of these ecosystems, but also the ecological and evolutionary dynamics of the organisms inhabiting them. Thus, rivers are biodiverse because they themselves are diverse – and they provide a wide variety of services and functions which add value to society and the environment, such as amenities, water supply and flood regulation (Southwood 1977, Harper and Everard, 1988, Palmer et al., 2010, Ekka et al., 2020, Hildrew and Giller, 2023).

River ecosystems contribute to the overall biodiversity of the river network, and broader freshwater ecosystem, by providing a diverse array of habitats (MacArthur 1965, MacArther and Wilson, 1967, Hynes, 1970, Tockner and Ward 1999, Vaughan et al., 2009). This is due to their interconnectedness with the terrestrial environment as they flow from their source, the headwaters (streams), towards the sea. The role of rivers in connecting habitats is therefore critical for maintaining a balanced and sustainable environment, supporting biodiverse communities including benthic communities, higher plants and fish (Townsend and Hildrew, 1994). The success of these instream species and many others, such as waterways birds, which occupy riparian habitats and feed on aquatic insects, is dependent on the broad range of habitats that rivers support. For example, dippers (*Cinclus cinclus*) feed on the larvae of caddis flies (order Trichoptera) and mayflies (order Ephemeroptera) which live in streams and rivers. The river network also provides habitats for several species of European importance. These include the kingfisher (*Alcedo atthis*), listed in Annex I of the European Union (EU) Birds Directive (BD), the freshwater pearl mussel (*Margaritifera margaritifera*), the white-clawed crayfish (*Austropotamobius pallipes*), the otter (*Lutra lutra*) and fish species such as the sea lamprey (*Petromyzon* *marinus*), brook lamprey (*Lampetra planeri*) and Atlantic salmon (*Salmo salar*), listed on Annex II of the Habitats Directive (HaD).

The definition put forward by the Convention on Biological Diversity adopted in 1992 (Secretariat of the Convention on Biological Diversity, 2011) recognises the importance of this link between organisms, their communities and the habitats which support them, defining biodiversity as:

> *'The variability among living organisms from all sources including, inter-alia, terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are part; this includes diversity within species, between species and of ecosystems.'*

For the purposes of this report, biodiversity is simply defined as:

> *'The diversity within species, between species and of ecosystems.'*

Rivers are intricately linked with the adjacent landscape and are therefore sensitive to how land is managed (Coffey et al., 2016, Duffy et al., 2020, Crooks et al., 2021, Petsch et al., 2021). The exploitation of rivers for society's needs has, however, led to the widespread degradation of their natural character, resulting in a loss of biodiversity, habitat attributes, and ecosystem services (Ward et al., 1999). This has arisen from catchment-scale land use changes such as urban development, intensive agriculture and forestry. Amongst these pressures is nutrient enrichment – a longstanding environment pollutant (Foy et al., 2003) which is typically associated with a failure to achieve good water quality and diverse ecological communities across NI.

Over the past two decades, there has been increased attention on the physical damage caused by hydromorphological pressures to the river network. These pressures include channelisation, arterial drainage, poor riparian management and physical barriers arising from two primary drivers: agricultural land use and urbanisation. These impacts are significant as they can degrade instream habitats, affect biodiversity, and diminish the overall integrity of river habitats. Hydromorphological pressures have also been linked to reduced water quality and altered hydrological regimes, which directly impact species that depend on natural flow patterns and intact physical habitats. This focus has also placed renewed emphasis on the critical link between species and their habitats and how both nutrient enrichment and habitat modification act individually or together as local drivers of biodiversity variability. Together with global drivers such as climate, the result is that river ecosystems are among the most threatened freshwater habitat types and are at risk from future climate and land use change (Coffey et al., 2016). The sustainability of river environments and their related diversity of communities is therefore very much under threat. Efforts to mitigate these impacts focus on river restoration, through removing barriers to connectivity, and implementing more sustainable water and land management practices to protect biodiversity and ecosystem services.

Further knowledge of the ecological responses to changes in the riverine environment is urgently required for informed and robust conservation. This requires an acknowledgement of the holistic nature of rivers, integrating sound scientific principles with management perspectives and policy ambitions which promote sustainable environmental heterogeneity. Considered individually irrelevant in nature, the accumulative impact of small streams to habitat provision, water quality and riverine biodiversity must also be given greater consideration. Moreover, there remains a critical need to evidence the impact of drivers of biodiversity loss. This includes pressures of nutrient enrichment and habitat impairment across the river network and how such relationships may change with increased conservation effort in the presence of multiple and interacting pressures.

Chapter 2: Biodiversity patterns across the riverine landscape

The protection of river ecosystems through appropriate management is fundamental to having a diverse species pool. Effective conservation strategies are therefore essential to reverse trends in biodiversity loss and improve the state of river ecosystems. To be effective, the design and implementation of biological monitoring programmes and conservation measures should be based on a solid conceptual foundation and an all-inclusive understanding of the river ecosystem. They should also be supported by empirical evidence of patterns in biodiversity, together with the main drivers structuring these. This requires understanding of the organisation of stream benthic communities and examination of the prevailing structuring factors at multiple spatial scales.

The River Continuum Concept (RCC) provides the foundational principals to river ecology (Vannote et al., 1980), modelling broadscale spatial structure and function patterns in streams in terms of gradual and predictable change from headwaters to larger rivers. It illustrates that the accumulation of discharge from headwaters to large rivers is accompanied by a decreasing topographic gradient, along with increasing depth and width. There are also changes from coarser to finer substrate, increased openness of the riparian canopy and alterations in trophic structure driven by allochthonous inputs to autotrophic production. This illustrates the dynamic equilibrium of river channel form driven by the dynamics of water and sediment discharge.

The RCC also emphasises the relationship between biodiversity and stream size, which is a key predictor of biodiversity in riverine systems (Heino et al., 2005, Pease et al., 2015). Typically, this relationship manifests as a relatively simple benthic trophic structure in the headwaters, with increasing complexity observed in the community structure in the downstream reaches. The RCC thus provides a linear perspective on riverine biodiversity, with headwater biodiversity relatively depauperate, with little contribution to whole stream biodiversity. This has contributed to headwater habitats, which constitute the majority of the river length in each catchment, being under-presented in biomonitoring and policy programmes with respect to their significance to the biodiversity m (Biggs et al., 2017). For example, in Ireland ecological monitoring stations are typically located at a minimum third order rivers, neglecting first and second order streams which, for Ireland, comprise 77% of the 74,000km of river channel (McGarrigle, 2014). Despite this, it is now understood that headwaters are the essential foundation for naturally functioning rivers. Specifically, they play an essential role in providing natural flood control, trapping sediments, retaining and processing nutrients as well as maintaining biological diversity which extends into downstream reaches, lakes and estuaries (Alexander et al., 2007). This biodiversity function provided by headwaters is afforded through their moderate variations in flow, temperature and discharge, which creates a unique physical template and facilitates their biodiverse character (Richardson, 2019).

Besemer et al. (2013) investigated the role of the dendritic structure of reaches, challenging the concept that biological diversity accumulated downstream. Their organism of study was river biofilms, the dominant mode of microbial life form in the river ecosystem, which is critical to their function. This study demonstrated that microbial diversity decreased from headwaters to downstream reaches. In addition, there was a high degree of variability in species composition among headwater streams that could not be explained by geographical distance between catchments. This suggests that the dendritic nature of the fluvial network constrains the distributional patterns of microbial diversity. Thus, the importance of the smaller streams within the headwaters of catchments and their role in the maintenance of microbial biodiversity within the river network is highlighted. Moreover, further studies in Ireland have shown that 29% of a catchment's macroinvertebrate diversity is unique to the headwaters (Feeley and Kelly-Quinn 2012, Callanan et al., 2014).

Headwaters also support many species common to both upstream and downstream reaches, highlighting their potential to act as biological sources to downstream sites if natural or anthropogenic pressures cause local extinctions (Feeley and Kelly-Quinn, 2012, Callanan et al., 2014). Meyer et al. (2007) categorised the biodiversity of headwater streams into species that are unique to the headwaters of river networks and to species which occur within the headwaters but also in downstream reaches. There are also species which move into the headwaters seasonally as well as those which migrate there to complete a particular life history stage. For example, native resident fish species such as brown trout (*Salmo trutta*) as well as migratory fish populations such as Atlantic salmon (*Salmo salar*) and river lamprey (*Lampetra fluviatilis*) rely on the upstream migration of adult individuals to maintain their juvenile populations. Headwater habitats also host other species of conservation concern. These include the European eel (*Anguila anguilla*), brook lamprey (Lampetra planeri), freshwater pearl mussel (*Margaritifera margaritifera*), white-clawed crayfish (*Austropotamobius pallipes*), the otter (*Lutra lutra*) and kingfisher (*Alcedo atthis*). The biodiversity value of headwater environments also includes species which live near these streams in semi-aquatic or riparian habitats. For example, riparian habitats are important for several bird species, such as the grey wagtail (*Motacilla cinerea*) and the dipper (*Cinclus cinclus*), which is commonly found near fast-flowing upland rivers, where it feeds and nests.

While headwater habitats are important for rare or protected species, there are an array of populations which make a significant contribution to either regional or catchment biodiversity and, in turn, to the ecological integrity of the entire river network (Furse, 2000, Heino et al., 2005, Clarke et al., 2008, Finn et al., 2011). However, in many instances the biodiversity value of headwaters is underestimated or unknown with respect to populations present, as there is not enough of specialised species identification to determine their full biodiversity potential. For example, for headwaters of the Upper Bann catchment the first recording of the mayfly species, *Baetis altanticus*, was documented as part of the long-term freshwater ecological programme undertaken by Catchments Unit at the Agri-food and Bioscience Institute (AFBI; Snounou, Snell and Feeley 2022). To give a brief history, *Baetis altanticus* was first discovered in England in 2017 and confirmed in Ireland in 2019. This is the first new baetid species discovered and confirmed in Ireland in more than 120 years (Feeley and Macadam, 2020). However, *Baetis altanticus* was only discovered in NI because of a long-term high resolution biomonitoring programme undertaken by AFBI. This highlights the significance of dedicated ecological datasets in revealing the complete biodiversity potential of the river network.

Given the ongoing declines in water and habitat quality across NI river networks (see Chapter 6), it is essential to reassess the significance of the combined total of the dendritic nature of the small stream network (SSN) to the biodiversity potential of each headwater branch. It should be noted that there is no universally accepted definition of what constitutes a small stream, and the term is often used interchangeably with headwater. However, here, the SSN includes headwaters defined on the Strahler stream order of less than or equal to second order and, while most small streams lie in headwater reaches, they also include small lowland and short coastal streams (Ovenden and Gregory, 1980, Moore and Richardson, 2003). It is important to note that the SSN can also be characterised from source (up to 2.5km; Furse, 2000) or by its catchment area (up to $10km^2$; Gomi et al., 2002).

Regardless of the definition, it is now becoming widely recognised that these streams play a crucial role in terms of habitat, refugia and reservoirs, significantly influencing downstream water quality (Richardson, 2019). It is essential to reevaluate headwater streams and the role of SSN in combating biodiversity loss within freshwater systems. Additionally, compared to downstream reaches, the SSN is crucial in assessing the effects of individual and combined drivers of change within the river network. Downstream reaches, due to their size and increasing number of pressures present, have greater potential to confound the identification of nutrient and habitat impacts on biodiversity. For instance, when P affects instream biodiversity, the SSN may offer a clearer understanding of how agricultural land use influence biodiversity patterns, as opposed to larger streams that receive substantial P discharges from various sources. Therefore, this review will showcase examples of how nutrient enrichment and hydromorphology impact biodiversity across the river network and within the SSN.

Another important consideration is that the biodiversity observed for the SSN can extend up into the small stream drainage ditches, as demonstrated by an intensive agriculture catchment in the south-east of Ireland. Indeed, Kavanagh and Harrison (2014) found that such habitats can support a diverse and species-rich community, including nationally rare and threatened species. Moreover, drainage ditches are a common feature of the Irish landscape and those associated with coastal and floodplain grazing marsh have high conservation value. The importance of this habitat for rare and threatened species is recognised with priority status under the United Kingdom's (UK) Biodiversity Action Plan (BAP). Clarke (2015) conducted an analysis of the current condition of Sites of Scientific Interest (SSSI) for the wider coastal and floodplain grazing marsh habitat, demonstrating that pressures affecting these sites included eutrophication and non-native species. This is outside the scope of this review but, building on the complexity of the general RCC concept, it also highlights the interconnectedness of pressures across the landscape.

Chapter 3: The policy context for biodiversity protection

From a global conservation policy perspective, the first coordinated effort to safeguard biodiversity was realised through the Ramsar Convention on Wetlands in 1971 which focused on the sustainable management of wetland habitats. In 1992, the Convention on Biological Diversity (CBD) introduced a global framework to promote biodiversity conservation. The CBD Strategic Plan for Biodiversity 2011-2020 (Convention on Biological Diversity, 2010) then led to the development of 20 Aichi Biodiversity Targets. These targets included specific reference to the need to conserve rivers (Target 11) and reduce both habitat loss (Target 5) and pollution pressures (Target 8). More recently, the Sustainable Development Agenda for 2030 integrates 17 Sustainable Development Goals (SDGs) adopted by the United Nations (UN) in 2015. These SDGs guide both national and international efforts in biodiversity conservation.

In the context of European freshwater conservation policy, protection for biodiversity and actions to halt the loss of aquatic biodiversity is presented in various EU strategies (e.g. the EU Green Deal, Farm to Fork and Biodiversity Strategy for 2030). Together, the EU Biodiversity and Farm to Fork strategies aim to reduce nutrient losses from agriculture by at least 50% by 2030 while also maintaining soil fertility. Moreover, the EU Biodiversity Strategy for 2030 provides an overarching framework, interlinking with the international CBD, Aichi Biodiversity Targets and the UN's SDGs. Thus, these overarching policy frameworks seek to protect and, where necessary, restore rivers by conserving or re-establishing biological communities which approach conditions found in unaltered river systems.

Within the EU, the legislative framework to protect and restore biodiversity in rivers is linked through interdependent European nature conservation and water management policies. This includes the EU HaD (92/43/EEC), the BD (79/409/EC) and the Water Framework Directive (WFD: 2000/60/EC). The HaD, together with the BD, underpin an EU-wide Natura 2000 network of nature protection areas established to ensure the long-term survival of Europe's most valuable and threatened species and habitats. Specifically, the Natura 2000 includes the best examples of habitats and species, comprising Special Areas of Conservation (SACs) and Special Protection Areas (SPAs) – designated under the

HaD and BD respectively. This is supported by BAPs and the network of SSSIs and Areas of Special Scientific Interest (ASSIs: as defined under the Environment Order (Northern Ireland) 2002)). Within the National BAP for NI there is a continuing commitment to protect biodiversity through a series of targeted strategies and actions. These initiatives are designed to meet national and international biodiversity targets, ensuring sustainable management of natural resources while addressing threats like habitat loss, nutrient pollution, and climate change. The NBAP forms part of the broader UK Biodiversity Framework, with specific objectives tailored to Northern Ireland's unique environment. The European Red List of Threatened Species, meanwhile, provides assessment and listings of conservation status for European species, with the production of Red Lists being a required action under the National BAPs.

The quality of rivers with respect to nutrient concentrations is an objective of a range of European policies. These all converge to the singular target of achieving at least good water quality by 2027, as defined under an overarching Integrated Water Resource Management (IWRM) framework presented by the WFD. The overall aim of the WFD is to achieve 'Good Ecological Status' (GES) for all water bodies based upon biological, chemical and habitat quality. This is supported by directives such as the provision of Urban Wastewater Treatment Directive (UWWTD). This directive aims to protect the environment from the adverse effects of wastewater discharges, particularly in urban areas. The control of discharges from wastewater treatment works (WWTWs) is an important part of this directive. By regulating the quality of treated water released into rivers, these measures contribute to the broader goals of conserving biodiversity and preventing damage habitats and communities from nutrient pollution.

The WFD is reinforced by the Nitrates Directive (91/676/EEC), which aims to reduce nitrate pollution from agriculture and promote good farming practices. In NI, the WFD is further strengthened by directives on Drinking Water (98/83/EC), Bathing Water (2006/7/EC), Integrated Pollution Prevention and Control Directive (2008/1/EC) and Groundwater (2006/118/EC). These directives play a critical role in supporting river biodiversity by protecting water quality and ensuring sustainable agricultural and industrial practices. The Water (Amendment) (NI) (EU Exit) Regulations 2019 ensure the continued implementation of the WFD and related water legislation post-Brexit.

Through the IWRM framework, the WFD advances biodiversity protection by promoting the sustainable management of water bodies, aiming for "GES" of rivers, lakes, and other aquatic ecosystems. Recognizing the role of physical processes like flow, sediment dynamics, and water chemistry in shaping river habitats—and acknowledging the impact of channel modifications and engineering practices—the WFD introduced the concept of "hydromorphology" (CEN, 2004). This term emphasizes the importance of managing both the physical and chemical aspects of rivers to support healthy, diverse ecosystems. The WFD also introduced specific designations and targets for Highly Modified Water Bodies (HMWBs), which are habitats irreversibly modified for human needs. Instead of aiming for GES, the goal for these habitats is to achieve "Good Ecological Potential", ensuring ecological improvements without compromising their human functions, such as flood defence or navigation, or without jeopardising the goals of the HaD (Hering et al., 2010). Overall, this approach supports biodiversity by improving habitat conditions while addressing challenges like invasive species management and maintaining the balance between ecological targets and human needs.

In addition, the WFD emphasises the importance of environmental flows (e-flows), the practice of using flow response relationships and societal water management goals to outlined sustainable scenarios for river flow regimes (Acreman and Ferguson, 2010, Horne et al., 2017). The WFD specifically defines this as "*a hydrological regime consistent with the achievement of the environmental objectives of the WFD in natural surface water bodies*," (Article 4, WFD). An EU-wide e-flow expert group also developed guidance on e-flows and with consideration to the HaD (WFD, 2014). Furthermore, the WFD also extends to nature conservation with specific reference, for example, to the management objectives for Natura 2000 sites.

The WFD addresses the ecological problems associated with the pressures of multipurpose water usage through a comprehensive monitoring programme. This affords protection to the ecological status of water bodies and opportunities for efficient water resource management to allow for the improvement of deteriorated ecosystems. Determining ecological status under the WFD relies on indicator organisms to assess different pressures of stream ecosystem. This includes both plant (macrophyte and phytobenthos) and animal components (benthic invertebrates and fish) which are combined for the classification of ecological status. Most importantly in terms of biodiversity conservation, when such monitoring programmes are supported by broader complementary data on land use (e.g. urbanisation, wastewater, agriculture, forestry) and human activities (e.g. water abstraction, recreational activities) within a river catchment, this has the potential to provide a step change towards finding targeted measures balancing the numerous and often co-occurring pressures rivers are subjected to and which threaten the biodiverse character of the river network.

In conclusion, effective river biodiversity conservation relies on maintaining functional river processes, such as natural flow and sediment dynamics, which support diverse biological communities similar to those in undisturbed – 'natura' - environments. Under the WFD, scientific literature and monitoring frameworks emphasize the importance of protecting these processes to ensure ecological health. In Northern Ireland, policy plays a crucial role in safeguarding river biodiversity by aligning local and international strategies and initiatives with WFD requirements, addressing issues such as habitat modification, nutrient enrichment and sustainable water use. These policies collectively work to balance ecological integrity with human needs, ensuring biodiverse ecosystems.

Chapter 4: The major drivers of biodiversity

The channel form provides the physical framework that allows for the biodiverse character of rivers (Hynes, 1970, Southwood, 1977, Townsend and Hildrew, 1994). However, channel form is commonly subject to modification by anthropogenic pressures such as the construction of physical barriers, which directly influences channel dynamics (Malmqvist and Rundle, 2002, Vaughan et al., 2007). The flow environment is also increasingly subject to changes in the frequency of hydrological extremes due to climate change, changes in soil temperature and increased hard surface areas within catchments from urban development (e.g. Ockenden et al., 2016, Miller and Hutchins, 2017, Mellader and Jordan, 2021, MacKenzie et al., 2002). Subsequent changes in flow directly impact the input of nutrients, primarily P, which can result in rivers becoming excessively loaded with nutrients (Miller and Hutchins, 2017, Ockenden et al., 2017, Forber et al., 2018). Examples of sources include fertiliser applied to crops and grassland being washed into rivers during rainfall, or wastewater discharges from sewage treatment plants, farmyards and domestic dwellings. This increases the potential for eutrophication, a long-standing environmental issue within NI rivers and broader freshwater systems including Lough Neagh, into which six main rivers flow (Elliott et al., 2016, McElarney et al., 2021).

Nutrient enrichment and hydromorphology are two key factors exerting significant pressure on river water bodies in Northern Ireland. Specifically, the impacts of nutrient enrichment are highlighted in terms of P, which is the primary limiting nutrient and pollutant in rivers. Pressures in P are typically attributed to diffuse sources from agriculture and point sources such as farmyards and discharges connected to sewage treatment plants and septic tanks. Hydromorphology is subsequently discussed in terms of its three defining elements under the WFD namely: morphological condition, river continuity and hydrological regime.

I. Nutrient enrichment as a driver of biodiversity

Nutrients (N and P) are important for organism growth and energy transfer in aquatic ecosystems. However, elevated levels of these nutrients often promote algal growth and produce shifts in community composition. For example, while typically associated with coastal areas, N can contribute to freshwater eutrophication – and both dissolved inorganic nitrogen and dissolved reactive phosphorus are limiting to the growth of algae, macrophytes and invertebrates (Camargo and Alonso, 2006, Ferreira et al., 2015). However, P is commonly the limiting nutrient for primary production in freshwater ecosystems (Schindler et al., 2008, Elser, 2012), and elevated concentrations in rivers stimulate plant growth (Mainstone and Parr, 2002). This in impacts habitat conditions (e.g. depletes oxygen levels, changing pH), affecting species like macroinvertebrates and fish (Smith, 2006, Hilton et al., 2006, Smith and Schlinder, 2009, Dodds and Smith, 2016).

The relative influence of P and N in river eutrophication processes is complex (Smith and Schindler, 2009), driven by seasonal periods of limitation and biological sensitivity. Moreover, nutrient additions, particularly from agricultural and urban sources, can be moderated by nutrient-limiting conditions, such as low availability of either N or P. This has been shown to cause a disturbance in the balance of N:P, modifying their ratio within freshwater ecosystems. For example, Blass and Kroeze (2016) cite agricultural practices and sewage as the main causes of nutrients entering European N-limited rivers. In P-limited rivers, sewage is the dominant source of P, except for those rivers draining into the Atlantic Ocean, where agriculture can also be dominant. In general, and for NI, the main nutrient pressure for rivers arises from P addition but additions of N or changes in the ratio of N:P can also significantly impact the biodiversity of river systems (Mainstone and Parr, 2002). Understanding catchment scale dynamics of N:P ratios is therefore essential for effective nutrient management strategies aimed at restoring and maintaining biodiverse river ecosystems (Smith et al., 1999; Conley et al., 2009). Studies have highlighted that the impact of imbalanced nutrient input - nitrogen levels that rise disproportionately compared to phosphorus, or vice versa – depends on local conditions and land use practices (Carpenter et al., 1998; Howarth et al., 2000). Additionally, climate factors, such as changes to temperature and precipitation patterns, can influence nutrient runoff and cycling, further affecting the N:P ratio. Overall, such imbalances can favour the growth of certain algal species, leading to harmful algal blooms that disrupt food webs and degrade water quality. Therefore, while eutrophication might be managed by controlling P alone and forms the central focus of this review, a more detailed evaluation of the respective roles of these nutrients, and their interactions, is required. This will aid understanding on how to mitigate the effects of both N and P to manage their adverse impacts upon the environment, especially in the presence of multiple and emerging pressures, including shifts in weather patterns driven by climate change.

Nutrient enrichment, commonly referred to as eutrophication, is amongst the most significant drivers of the global decline in aquatic biodiversity (Smith, 2003). Specifically, eutrophication relates to the excessive input of N and P into rivers and lakes (Dudgeon et al., 2006, Vörösmarty et al., 2010), and is the process though which water bodies become overly enriched with nutrients leading to the excessive growth of algae and other aquatic plants resulting in lower oxygen levels and increased turbidity (Hilton et al., 2006, Withers et al., 2014). This can create changes in community composition, reducing the richness of species within rivers and, ultimately, impacting negatively upon biodiversity. For example, when exposed to initial nutrient pressure, macroinvertebrate communities which typically consist of small numbers of multiple species start to change into communities containing fewer species in much larger numbers. If nutrient enrichment persists, this can manifest in the community as the progressive disappearance of indicator species, until few remain and the community is dominated by species which were either previously present in low abundance or not present at all (Hellawell, 1986). Such population changes also directly impact any surviving fish which feed on the macroinvertebrate communities (Thera et al., 2020).

The occurrence of nutrient enrichment has been increasing over the last decade, primarily due to agricultural intensification, forestry practices and wastewater treatment processes, which has implications for the provision of ecosystem services such as drinking water, fisheries and recreation opportunities. This poses a serious threat to biodiversity, affecting both the river ecosystem and the wider services it provides. Moreover, the input of nutrients into rivers typically occurs in an infrequent but continuous manner, subject to catchment characteristics and prevailing land management practices. Addressing nutrient management is therefore a key policy objective for preserving NI’s rich biodiversity. Consequently, definitions have been put forward through the UWWTD and NiD pertaining to the river habitat as outlined below.

The UWWTD defines eutrophication as:

> *"The enrichment of water by nutrients, especially compounds of nitrogen and/or phosphorus, causing an accelerated growth of algae and higher forms of plant life to produce an undesirable disturbance to the balance of organisms present in the water and to the quality of the water concerned."*

The NiD defines eutrophication as:

> *"The enrichment of water by nitrogen compounds, causing an accelerated growth of algae and higher forms of plant life to produce an undesirable disturbance to the balance of organisms present in the water and to the quality of the water concerned."*

Despite, nutrient enrichment being the focus of research and policy objectives, with much known about the general effects on river ecosystems, substantial knowledge gaps remain regarding its impacts at local and catchment level. Moreover, there is a compliancy gap with respect to the implements of environmental standards for nutrients and management of eutrophic risk. This can be attributed to the failure of current catchment management interventions to adequately address the diffusivity of nutrient sources and pathways (Thomas et al., 2021). Understanding how nutrient concentrations and loads may change due to urban development and agricultural land use change is a key step in developing a greater understanding of future eutrophication risk and for designing and targeting management solutions. This should be approached at the catchment level, as environmental risk-mapping of pollutants in the Irish context — and extending to NI — has demonstrated that adopting a local scale catchment-specific approach is critical to effectively reduce N and P loads to rivers (Thomas et al., 2021, Service 2024). This will also be important in providing specific insights into how nutrient dynamics operate in different contexts are necessary to support such management approaches and for biodiversity protection.

II. Hydromorphology as a driver of biodiversity

Hydromorphology pertains to the physical state of freshwater systems, which creates habitats and supports natural processes essential for sustaining diverse ecological communities. Examples of these natural processes include sediment transport, flow dynamics, and the seasonal flooding of riverbanks, all of which contribute to habitat diversity and ecosystem functioning. The main hydromorphology drivers for river biodiversity include flow variability, channel structure, sediment composition and either the loss of, or increased, connectivity to surrounding ecosystems. These factors influence habitat availability, species distribution, and riparian communities, ultimately determining the functionality of riverine ecosystems. This can arise from management practices such as straightening, widening, deepening or dredging channels as well as removing riparian vegetation, land drainage, installing flood protection or other physical structures and urban development. This has direct consequences for the biodiversity of rivers, especially if modification occurs within more sensitive habitat areas such as the headwaters that are used for spawning (Atkinson et al., 2020). Moreover, while silt plays an important role in supporting such habitats, excessive levels can be detrimental to species across all trophic elements and particularly if it is associated with nutrient pressures (Davis et al., 2018).

Thus, as hydromorphology is a broad term that encompasses various drivers, the WFD identifies three key hydromorphological elements related to rivers. These elements will be used here to illustrate their impact on river integrity and biodiversity. They include:

I. ***Morphological condition*:** Channel features of depth, width, substrate and the riparian zone.
II. ***River continuity*:** Connectivity of the river network.
III. ***Hydrological regime*:** Quality of flow and connection to groundwater.

I. Morphological condition

River biodiversity is linked to the diverse hydrogeomorphic patterns throughout the river network (Vannote et al., 1980). These patterns arise from the shape and size of the channel, the distribution of riffle and pool habitats, the abundance and diversity of gravel bars and the stability of the substrate, determined by the interactions between flow regime and local geology and topography (Frissell et al., 1986). These processes are underpinned by the dynamic transport of water and sediments which vary spatiotemporally across the catchment, shaping channel form, complexity and connectivity (Southwood, 1977). The high diversity of sediment size is therefore critical to supporting diverse instream communities because patches differing in grain size often constitute different functional habitats (Harper et al., 1992). For example, macroinvertebrate diversity, abundance, traits or productivity have been shown to demonstrate dependencies with substrate diversity and surface perimeter ratio (Beisel et al., 2000).

While sedimentation is a naturally occurring phenomenon in rivers, land-use changes have resulted in an increase in anthropogenically-induced fine sediment deposition. The term ‘fine sediment’ used herein refers to sediment less than 2mm in size. Collectively, these sediments are referred to as siltation and when present in sufficient quantities contribute to a decline in habitat condition and instream communities (Berkman and Rabeni, 1987). River depth and width are also key features of morphological condition and in a natural river system are adjusted to the flow regime. Modification of this equilibrium can contribute to the erosion of bed and bank material and siltation of the riverbed. In particular, it decreases instream heterogeneity, alters sediment dynamics and changes the composition of shallow riffles and gravel bars (Wood and Armitage, 1997).

The riparian zone, located between adjacent terrestrial and aquatic ecosystems, is also a key feature of the morphological condition. It encompasses a diverse suite of ecosystem types, from headwaters to downstream reaches, including riverbanks, floodplains and wetlands. Riparian zones play a key role, via their vegetation structure, in determining stream biodiversity as well as stream processing and function. For example, riparian vegetation is important in regulating instream microclimate, water quality and temperature, preventing riverbank erosion and promoting bank stability. In addition to being an important component of river corridor biodiversity, riparian vegetation communities also help to reinforce riverbanks with their robust root systems, making them more stable. Simplification of the river corridor due to pressures such as channelisation and inappropriate farm practices like grazing, however, has contributed to habitat degradation and negatively impacted instream communities (Peipoch et al., 2015, Wohl, 2019).

II. River continuity

River ecosystems are integral features of the landscape, shaped by the transport of water and materials from their surrounding catchment to river network (Hynes, 1975). A key driver of pressures on river continuity in NI relates to the installation of physical barriers and other similar types of engineering works (e.g. roads, bridges and dams). Such modifications disrupt the natural flow and result in habitat fragmentation, which significantly impact biodiversity by disrupting the movement and life cycles of aquatic organisms. Hydrological alterations, such as water abstraction and flood control measures, further strain river continuity. Additionally, pollution from agricultural runoff and industrial, urban and wastewater discharges, along with the spread of invasive species, exacerbates these pressures, collectively threatening aquatic biodiversity.

Although recognised as being an important ecosystem aspect, river continuity is primarily a hydromorphological attribute that ensures the free movement of species, nutrients and sediment. This continuity supports species migration, reproduction, and access to vital habitats. Thus, defining continuity and understanding all of this core components is critical to understanding the fragmentation of river habitat and the key factors which that break river continuity – such as physical barriers and land use changes. Moreover, due to the likely significant impact of barriers on Northern Ireland's rivers and the EU target to restore at least 25,000 km of river by 2030, understanding river connectivity is essential for developing effective mitigation strategies to restore habitat, reduce degradation, and support the recovery of river ecosystems.

From the perspective of landscape ecology, Ward (1997) defined riverine connectivity as '*energy transmission across a river landscape*'. Based on this concept, Pringle (2001; 2003) broadly defined hydrologic connectivity from the perspective of freshwater ecology as '*the water-mediated transfer of matter, energy and organisms within or between elements of the hydrologic cycle*'. This continuity of water flow and the spatial connection across the catchment and within a river channel network is critical to ensuring diversity of instream habitat (Freeman et al., 2007). Here within this report, river continuity is considered in terms of the ability of water, sediment and aquatic species to move freely within the river corridor and across the catchment. This movement is typically described in four dimensions – longitudinal (upstream and downstream in the river channel, including to estuarine and ocean systems), lateral (between the main channel, riparian areas, floodplain, and catchment), vertical (between groundwater, river interstitial zone and atmosphere) and temporal (natural flows that include seasonal variations; Ward, 1989).

Longitudinal connectivity holds a temporal dimension which is required to support the complex life cycles of many aquatic species. Ecological processes, such as reproductive cycles of macroinvertebrates and fish, are driven by seasonal changes in flow and the frequency of flooding (Parasiewicz et al., 2019). Interrupting these cycles, especially for fish species which depend on flow conditions and their ability to move freely through the stream, has a dramatic impact on successful spawning and recruitment processes, which are temporal events (Warren et al., 2015). Temporal connectivity is crucial for identifying key spatial connections across catchments, particularly for sediment and nutrient flow, and will become increasingly important under climate change predictions.

III. Hydrological regime

In the riverine system, the complex interaction between the natural flow regime and physical habitat is a major determinant for maintaining diverse communities for multiple organism groups including plants, phytobenthos, macroinvertebrates and fish (Poff and Allan, 1995, Ward et al., 1999, Ward et al., 2002, Lytle and Poff, 2004). It is not only the quantity of discharge that is important for sustaining and conserving species diversity but also, the timing and flow dynamics (Poff et al., 1997, Bunn and Arthington, 2002). Local scale instream variations in flow and near-bed velocities can dictate the distribution and abundance of particular species of plants and animals. For example, changes in rates of water level, fluctuation, disturbance frequency (floods and spates) and intensity (velocity and shear stress) can affect plant growth rates (Wetmore et al., 1990). facilitating connectivity and transporting materials and organisms. It also acts as a key agent of disturbance, shaping riverine environments (Sponseller et al., 2013). For example, high flow events, such as floods, can scour riverbeds, displace sediments, and alter habitat structures, which can be both destructive and regenerative (Bunn and Arthington, 2002). These disturbances can clear out accumulated organic matter and create new habitats such as gravel bars and side channels that benefit certain aquatic species. On the other hand, excessive disturbance from altered flow regimes—due to activities like damming or water abstraction—can destabilize habitats, disrupt species' life cycles, and reduce biodiversity. Flow disturbances are therefore essential for maintaining the dynamic equilibrium of river ecosystems, but when significantly altered, they can lead to habitat degradation and loss of species diversity.

There are four key principles which demonstrate the important mechanisms linking hydrology and aquatic biodiversity. These are presented here to illustrate how natural flow dynamics can also be impacted by other factors such as channelisation, drainage, urbanisation and physical barriers. Firstly, and as outlined within Resh et al. (1988), physical disturbance from flood events is a major determinant of the spatial and temporal dynamics of benthic communities. Secondly, aquatic species have evolved life history strategies, primarily in direct response to their natural flow regimes. Thirdly, the maintenance of longitudinal and lateral connectivity by natural flow dynamics is essential to the viability of populations of many riverine species (Resh et al., 1988; Bunn and Arington, 2002). For example, flow plays a critical role in the lives of fish, with life events linked to the flow regime (e.g. phenology of reproduction, spawning behaviour, larval survival, growth patterns and recruitment). In addition, many of these life events are synchronised with temperature and day-length such that changes in flow regime which are not in natural harmony with these seasonal cycles may have a negative impact on aquatic biota (Humphries et al., 1999). Fourthly, the invasion and success of introduced species in rivers is facilitated by the alteration of flow regimes (Resh et al., 1988; Bunn and

Arthington, 2002). Note, while important, it is considered that the connection of the riverine network to groundwater is beyond the scope of this review.

Despite recognition of these mechanistic relationships, predicting and quantifying the biotic responses to altered flow remains challenging due to the complexity of ecological interactions. Various drivers, such as water abstraction, agricultural and urban runoff, installing of flood measures such as damming and other physical barriers, and climate change, exert pressures on rivers by altering flow patterns, reducing water quality, and fragmenting habitats. The ecological responses to these pressures depend on multiple hydrological components, such as the frequency, duration, and intensity of flow changes. Local factors, including the specific hydroclimatic conditions, the biological traits of species, and how changes in flow affect stream hydraulics, further complicate predictions. In NI, information on these relationships is limited especially for individual catchments, and more research is needed to understand how altered flow regimes impact biodiversity and river ecosystems, which is critical for developing effective catchment management and restoration strategies.

Chapter 5: Evolution of key pressures

Nutrient enrichment, alongside a long history of physical alterations to river habitats, is significantly damaging river biodiversity. The pressure from P pollution has developed over time, exacerbated by agricultural runoff and wastewater discharges. Additionally, human exploitation of rivers — such as the construction of physical barriers like dams and weirs, along with arterial drainage — has disrupted natural flow and impaired instream habitats. These modifications also impact river continuity, negatively affecting the ecological status and biodiversity of river systems. Addressing these issues, and their development through time, is critical for the restoration and conservation of NI's river ecosystems. Therefore, evolution of these key pressures – phosphorus enrichment, physical barriers and arterial drainage – on NI's river ecosystems are presented below, as it provides valuable insights into their historical context and impact, enabling more effective management strategies and informed decisions to protect and restore biodiversity.

a. Nutrient enrichment

In NI, persistent nutrient pollution from human activities is one of the most significant long-term pressures on river systems and extending in the wider freshwater system as demonstrated by Lough Neagh (Cave and Allen, 2023). This growing issue is a major driver of habitat degradation and biodiversity loss nationally and globally (Dudgeon et al., 2006, Vörösmarty et al., 2010), leading to the proliferation of opportunistic algal blooms. These changes disrupt the natural balance of riverine food webs, further threatening the integrity and specifically ecosystem stability in NI. Addressing nutrient pollution is thus crucial for protecting riverine biodiversity.

I. Phosphorus enrichment

High P concentrations in NI rivers can be traced back to the introduction of P-based detergents and exponential population growth in the 1950s. This was followed by increased use of artificial P fertilisers, along with a surge in livestock populations to support ambitious food production targets in the 1980s (Environment Agency, 2019). This, together with effluent from wastewater treatment plants (WWTPs) (Jarvie et al., 2006), septic tank systems (STSs) (Withers et al., 2014, Gill et al., 2018) and industrial discharges (Richards et al., 2015b) has contributed to long-term and persistently high concentrations of P in rivers, exceeding levels typically observed under natural conditions (e.g. as per WFD).

Improving and protecting surface water quality has subsequently been at the forefront of environmental research and policy for many decades. This extends to environmental monitoring with ongoing monitoring Barry and Foy (2016) demonstrating how trends in nutrient pressure have changed over a 20-year period. Initially, the average P concentration in NI rivers decreased in the early 1990s, attributed to improvements in wastewater treatment processes, the reduction of P content in detergents (Foy, 2007) and response to agricultural policy (Barry and Foy, 2016). The introduction of Phosphorus (Use in Agriculture) Regulations (Northern Ireland) 2006 also led to a reduction in P from agricultural activities (Doody et al. 2020), in conjunction with ongoing improvements in domestic wastewater treatment through investment by Northern Ireland Water (NIW; NIWATER, 2024).

Despite the progress made in reducing discharges from industrial, agricultural, and municipal point sources, significant water quality problems persist from inadequately regulated sources, especially in rural areas. National monitoring data indicates that P continues to have a significant impact on water quality, with levels of soluble reactive P in 93 rivers monitored under the WFD surveillance monitoring programme increasing from 0.047 mg/L in 2012 to 0.073 mg/L in 2022 (NISRA, 2023). This trend highlights how the nature and relative contribution of sources may change over time. Variations in P concentrations across NI are attributed mainly to diffuse sources associated with overland and through flow from agricultural land (Macintosh et al., 2018), contaminated groundwater (Nijboer et al., 2004), together with more local discharges such from farmyards and wastewater management (Doody et al., 2020, NIWater, 2024). Moreover, this data suggests that most catchments across NI have not yet achieved limiting P concentrations, underscoring the need for further research to understand this issue and effectively mitigate its impacts.

The sources of P and their impact may manifest differently along the river network. According to Macintosh et al. (2011), within the SSN, STS may have a disproportionate impact on the low-flow P concentration of receiving rivers. This emphasis that nutrient pollution – and especially P – originates from a variety of sources and pathways and exists across different temporal and spatial scales. Further adding to the problem of managing P enrichment is that it is both a rural and an urban problem. Although some general principles apply across ecosystems, heterogeneity in the ecological response within and across types – and across time and space – together with the heterogeneity in the types and scalar patterns of nutrient inputs, results in significant differences in how nutrient enrichment manifests at local and national scale. Therefore, tailored advice and strategies is critical for biodiversity remediation. Moreover, ecosystem responses to nutrients are influenced by the presence and types of other pressures affecting water quality (e.g. pesticides, industrial chemicals, thermal pollution) and/or the state of the habitat condition and accumulative stress experienced by the ecological community present (e.g. diseases, fishing pressure). As a consequence, a system's approach to protecting and restoring ecosystems impacted by nutrients will likely require a two-step process of managing nutrient addition, followed by action to address other pressures.

b. Hydromorphology

Hydromorphology elements have been subject to severe anthropogenic alterations for decades and even centuries, which has had a serious negative impact on the functioning of aquatic ecosystems and water quality. Changes in river catchments have resulted mainly from flood prevention measures and the negative effects of settlements and agricultural production, which has led to the regulation of water. Specifically, rivers have been exploited for activities including navigation, water and food supply, waste disposal, flood defences and power generation. To enable this, modifications have been made to water infrastructure facilities serving agriculture and wider water management (e.g. weirs, dams, flood defences, land drainage). Consequently, straightened channels, the presence of physical barriers and the removal of riparian vegetation are common indicators of human modification to NI catchments.

Such interference to the river habitat, however, affects hydrology, continuity of water regime and sediment dynamics, which disturbs the overall ecological and chemical status (Grizzetti et al., 2017). Moreover, many of the present pressures on rivers demonstrates the influence of past land use management on present day biodiversity, which have resulted in the long-term reductions in habitat and species diversity. Progressive urbanisation and intensive agricultural enterprises continue to contribute to land use change, which in general have a negative hydromorphological effect.

In this context, the longstanding practice of arterial drainage, which generally enhances connectivity across catchments, in contrast to instream physical barriers that fragment river habitats and reduce connectivity are examined below. Together, these two modifications of NI's rivers pose the most significant ongoing threat to biodiversity from a hydromorphological perspective at both regional and local scales, contributing to the loss of the natural character of habitats.

I. Physical barriers

Rivers have been modified by physical barriers throughout human history. These include small low-head physical structures such as fish weirs (stone or wooden barriers constructed in rivers or estuaries to deflect the fish into a net or basket) which can be dated back to 6,100–5,700 BC (Historic England, 2018). Bridge aprons, locks, dams, culverts and fords are also a common feature of rivers across the island of Ireland. Note that, for this review, emphasis has been placed on low-head structures, excluding large reservoir and storage dams (>5m in height). Such weirs were built to harness the flow of rivers to power mills. Thus, physical barriers have historical and cultural importance, although many are no longer in use.

The key issue from an ecological perspective is that the construction of weirs and other in-channel structures alter the natural hydromorphological form of a channel. It impounds the water and reduces its velocity profile on the upstream side while, conversely, creating a steep transition of gradient and area of accelerated velocity in the immediate downstream. The result is a discontinuity of river flow, degraded habitats, and altered water levels. This fosters an environment which impedes the natural siltation process, directly impacting upon instream species (Gargan et al., 2011), and in particular the migration of fish (Atkinson et al., 2020). These impacts of barriers can be variable – from short delays to complete obstruction – depending on barrier type, hydraulic conditions, species swimming capabilities and timing of migration. The most pressing issues faced by catchment managers are complete barriers, which can reduce or fragment species distribution completely. This results in diminished populations which are increasingly genetically isolated and at greater risk of extinction (Junker et al., 2012, Coleman et al., 2018). However, while barriers present a serious threat to Irish fish populations, Krieg and Zenker (2020) suggest that modifications to existing barriers such as bridges, culverts and dams can have positive impacts on indigenous species like crayfish, against invasive organisms. This highlights the importance of local conversation measures and the role of adaptions of physical barriers when removal may not be possible, which address local needs to protect instream biodiversity.

Presently, disrupted river continuity and degraded ecosystem functioning as a result of physical barriers is a characteristic feature of rivers within NI, Ireland and across the EU (Kelly-Quinn et al., 2022, Parasiewicz et al., 2023). It a has been estimated that there are on average 0.74 barrier for kilometre of river in Europe, with a median distance of 108m between adjacent barriers (Belletti et al., 2020). For Ireland, Atkinson et al. (2020) reported an obstacle density ranging from 0.02 to 1.2 obstacles per river kilometre across 10 river catchments. Furthermore, data for Ireland from initiatives such as the Adaptive Management of Barriers in European Rivers (AMBER: consult amber.international) project and Ireland's National Barriers Programme (NBP) conducted by Inland Fisheries Ireland (IFI), has identified 73,376 potential barriers within the Irish river network (IFI, 2024). This suggests that the situation across EU is reflective of the status of Irish waters.

To do date, while existing elsewhere across the UK (Jones et al., 2019) similar river obstacle inventories for NI are lacking (The River Trust, 2024). This is despite the fact that NI rivers are fragmented and in need of either restoration or protection to prevent deterioration. There is also a lack of local case studies and coordinated governance across NI on barrier removal which recognises the critical importance to biodiversity of achieving free-flowing river systems. The EU Biodiversity Strategy also aims to restore at least 25,000 km of free-flowing rivers by 2030 by removing barriers and restoring floodplains and wetlands. As such, there is an urgent need for an enabling and coordinated policy framework in NI to improve understanding on the number, type and location of physical river barriers. This needs to be addressed as such river obstacle inventories are a necessary first step for making decisions on remediation measures for rivers (Atkinson et al., 2018). Moreover, the issue surrounding physical barriers represents a key national but also transboundary issue for water quality and biodiversity management across the island of Ireland.

The current EU policy framework reinforces the importance and urgency of restoring connectivity in European rivers, supported by various regulations, including the HaD, WFD and the EU Eel Regulations, and strategies (e.g. Biodiversity Strategy for 2030). For example, impaired passage is a significant issue for migratory fish species listed in the HaD, for example, Atlantic salmon (*Salmo salar*) and the sea lamprey (*Petromyzon marinus*) and river lamprey (*Lampetra fluviatilis*). The WFD identifies the importance of longitudinal connectivity to the character of rivers, both for upstream and downstream movement of aquatic organisms and for sediment transport and re-naturalisation of constrained rivers. Moreover, a high-status river with respect to continuity is defined under the WFD as being '*not disturbed by anthropogenic activities*' and thus facilitating the '*undisturbed migration of aquatic organism and sediment transport*'. However, despite this ambition of the WFD, barriers continue to constitute a significant pressure to circa 20% of EU surface water bodies and are one of the main reasons for rivers failing to reach GES. Moreover, it is the cumulative impact of a large number of river barriers on EU rivers which is cited as one of the leading causes of the more than 80% decline in freshwater biodiversity and the loss of 55% of monitored migratory fish populations (Birnie-Gauvin et al., 2018, Moberg and Singler, 2020). Therefore, restoring rivers and streams affected by disrupted continuity due to barriers is one of the main challenges that needs to be addressed to protect river biodiversity not just in NI but across the EU. However, national policy currently inhibits barrier removal in NI. The implementation of the WFD does not specifically legislate for barrier removal, while planning protections for weirs erected before 1860 prevent the removal of obsolete barriers (The Rivers Trust, 2024). This, therefore, highlights a compelling need for a strategic approach, underpinned by enabling legislation, for barrier removal and fish passage installation measures within NI. Furthermore, given that barriers are a significant transboundary issue affecting water quality and biodiversity loss, any strategies developed should also encompass the entire island of Ireland and form part of a wider consultation process on mitigation of this pressure for river ecosystems.

II. Arterial drainage

Inefficient natural land drainage, coupled with relatively high rainfall and the consequent frequent flooding of rivers presents a characteristic feature of the Irish landscape. Specifically, the topographical configuration is of a high maritime perimeter and a flat interior which causes many Irish rivers to have reduced gradients, resulting in short channel reaches typically connected by lakes. For NI the combination of relatively high annual rainfall, low evapotranspiration and extensive areas of impermeable soils means that poor drainage is a characteristic of the landscape. Consequently, across NI and the island of Ireland, there is a long history of land drainage and channel modification. Typically, two types of drainage carried out: arterial and field. Arterial drainage involves the artificial widening and deepening of the main channel of rivers and important tributaries to increase their effectiveness in draining catchments. Field drains, meanwhile, are concerned with the removal of surplus water from fields (Bruton and Convery, 1982)

Since 1947 arterial drainage (channelisation) of more than 6,000km of river channel has occurred in NI, at a regional density of 0.34 km^{-2}, higher than the regional densities across the UK and Ireland (Brookes, 1988). Consequently, arterial land drainage is a prominent characteristic feature of the agricultural landscape in NI, which was encouraged by policy initiatives for greater agriculture productivity and economic prosperity. The most intensive period of channel modification was from 1930 to 1990, driven by war-time demand for increased agricultural output. This was sustained by the Common Agricultural Policy (CAP), and specific funding for land drainage improvements, to ensure high rates of agricultural productivity. In addition to this, channel form was modified as part of an extensive land drainage scheme which required improvements to the efficiency of the river network and flood protection to confine high flows (Brookes, 1988, Sear et al., 2012, Robinson, 1990). For example, guidance under legislation set out in the Drainage (NI) order 1973, led to river channel modification, which was typically designed to accommodate a five-year flood in agricultural areas and/or to provide increased channel capacity for discharge from extensive field drainage schemes (Brookes, 1988).

Construction of these early arterial drainage channels involved major hard engineering featuring the widening, straightening and deepening of existing channels. This had an enormous impact on instream habitats, along with sediment, vegetation and species, which were completely removed in most cases. For example, peak suspended sediment loads were greatly increased and river habitat loss in important nursery grounds was substantial as a result of meander shortening. Riparian zones were also severely impacted due to spoil heaps and habitat simplification (Bruton and Convery, 1982, Brooker, 1985, Brookes, 1988, Robinson, 1990). For the most part, however, the impact on biodiversity impairment and loss was largely unquantified. Associated with these modifications was a general requirement to maintain the design conveyance or standard of flood protection. This often involved the removal of sediment accumulations, or the reinforcement of river channel banks and beds where erosion threatened land or specific structures, for example, river embankments. This was distinct from weed clearance and debris removal, which were also part of the maintenance process. Maintenance therefore resulted in a continual disturbance of the river channel, which often significantly modified and further impaired the channel form. In addition, these practices were supplemented with some direct channel modification undertaken to enhance instream habitat and morphological diversity through river rehabilitation. This presented a significant secondary pressure to riverine biodiversity (Bruton and Convery, 1982, Brooker 1985).

Presently, the rate of new land drainage has slowed significantly, with most modification now focused on urban flood protection and the maintenance of existing drainage and food alleviation schemes. However, riparian landowners continue to have responsibility for erosion control and maintenance of water courses, especially the enormous network of minor water courses and ditches that often form the headwaters of catchments. The national picture is therefore one of long-term intervention extending over much of the river network. Thus, the actual extent of river channel modification is likely to be greater than currently represented, as many interventions remain undocumented, and maintenance efforts are largely unquantified across the river network.

Chapter 6: Current trends in water quality monitoring

The data for classifying the ecological status of NI water bodies, along with the main pressures facing them, is collected as part of the WFD national monitoring programme. It suggests that the status of river habitats is deteriorating, putting biodiversity at risk.

Across NI there are 450 river and 21 lake waterbodies. Monitoring of these bodies has shown that since 2015 the ecological status of rivers has not demonstrated improvement. Specifically, in 2018, 31% of NI streams achieved an ecological status of 'good,' while the rest failed to achieve the environmental objective. This compares with 33% which were classified as 'good' or better in 2015. In a more recent assessment in 2021, 31% achieved at least GES but no rivers achieved an overall status of 'good' or 'high' with includes both the ecological and chemical status. These results indicate that the level of anthropogenic pressure on the ecological status of river ecosystems remains high, with excess nutrients, primarily P, underlining the lack of improvement observed in NI rivers since 2015 (Northern Ireland Audit Office, 2024). In comparison, for Ireland, the Environment Protection Agency (EPA) Water Quality Report 2023 (EPA, 2024) showed that 55% of rivers obtained satisfactory biological condition (equating to 1,309 river bodies). This means that the status is relatively unchanged since 2018 and there is no indication of overall improvement. Excess nutrients, N and P, primarily from agriculture and wastewater treatment activities are cited as the main pressure on the ecology of waters, alongside changes to flow and physical habitat condition (Trodd et al., 2021). For Ireland there is also concern regarding the mobility of 'high' status sites and their decline was highlighted within the 2018-2021 cycle of the River Basin Management Plan (RBMP; EPA Water Quality Report 2019). This led to the formation of the Blue Dot Catchments Programme to protect and restore 'high' status water bodies (LAWPRO, 2024). This decline in 'high' ecological water status in rivers is a trend reflected throughout the EU, but relationships between drivers, changes in water quality parameters, and the aquatic ecosystem is complex (Diamantini et al., 2018). Therefore, it is critical to investigate the factors which cause sites to lose 'high' status at national and local scale in order to undertake measures to protect and restore 'high' status water quality (O'Donoghue et al., 2022). Addressing these failures, and those more broadly observed across the island of Ireland, requires coordinated efforts in habitat restoration and both nutrient and water management.

Chapter 7: Evidence of key pressures

Rivers and their catchments are complex, dynamic and non-equilibrium systems, hosting an array of flora and fauna. Although the general functioning of these ecosystems is known, there is often less knowledge about the characteristics of individual reaches across catchments in NI, making it difficult to inform future local scale management decisions in the face of nutrient enrichment. For instance, rivers are becoming increasingly impacted by urbanisation and are facing more intensive land use management practices (e.g. forestry and agriculture), resulting in habitat modification and nutrient enrichment - the main drivers of river-related biodiversity loss in recent decades. The evidence presented within this section is a summary of the literature evidencing the impact of pressures associated with hydromorphology and nutrient enrichment by the major plant nutrients (N and P) on the characteristic flora and fauna of NI rivers.

While outside the scope of this review, it is important to consider how climate change may impact weather patterns. This includes changing rainfall and temperature patterns, which may interact to influence the influence of nutrient enrichment within the river network. For example, higher temperatures and nutrient enrichment increase primary productivity, which directly affects the dimensions of biodiversity, such as species richness. The typical and common covariance between climatic factors and human impacts can future complicate the evaluation of their independent roles in determining biodiversity patterns and its loss. While brief comment is provided on climate, to adequately elucidate, an additional review is needed to examine the effect of climate change and human impact on biodiversity patterns, and how nutrient enrichment and habitat modification may alter biodiversity under future climate scenarios.

Part I. Hydromorphology: Pressures and impacts

The structural diversity of river habitat is critical to the maintenance of diverse biological communities and is commonly described in terms of hydromorphology, which reflects the inseparable association of channel form and flow. Modification to the hydromorphological character of rivers refer to changes to the natural flow regime and the structure of the river habitat. This includes modification of bank structures, sediment composition, discharge regime, gradient and slope. Moreover, hydromorphological pressures extend to physical alterations of the riparian zones, water level and habitat connectivity. This generally arises land and water management practices. Such pressures are typically the consequence of human activities (drivers) in the catchment area, including flood defence structures, agriculture, fisheries and urban development. Therefore, catchment land use and management practices, both present and historical, are important in the context of hydromorphological quality. Note, there is evidence, that artificial structures among other types of hydromorphological alteration can lead to the creation of new habitats which can be exploited by invasive species (Snell and Irvine, 2013). This can cause the replacement of the original species with invasive or opportunistic species. In turn, while this may result no obvious changes in biodiversity, significant changes in the composition of species can be evident. The role of habitat and how it evolves with respect to physical, chemical and biological elements is therefore an important consideration, but outside the scope of this study.

The impact of pressures to the hydromorphological character of rivers typically results in three categories of change, which will form the focus of this review. Firstly, changes in erosion and sediment transport, secondly the interruption of river and habitat continuity and thirdly, changes to the dynamic of river flow. The consequences of such pressures cause both direct and indirect impacts on the characteristic fauna and flora found instream (Elosegi and Sabater, 2013). The ecological classification system required under the WFD recognises the importance of habitat and requires that there are no more than very minor human alterations to the hydromorphological quality elements. It specifically describes hydromorphological elements as 'supporting the biological elements' assessed based on criteria expressing morphological conditions, river continuity and the hydrological regime. In practice, this means assessing pressures and impacts on three habitat criteria:

I. Morphology (the physical habitat – substrate composition, width/depth variation, bed structure, banks and riparian zone).
II. Continuity (the ability of sediment and migratory species to pass freely up and down rivers. Also includes the impact of lateral connectivity with the floodplain and wider catchment).
III. Hydrological regime (the quantity and dynamics of flow. Also include connection to groundwater which is outside the scope of this review).

Using this framework, pressures and impacts are grouped according to these habitat criteria and discussed below; according to the key pressures identified:

I. Key pressures impacting river morphology

River channels are fundamentally conduits for water and sediment. The specific processes of water and sediment movement amongst components of river systems – from upstream to downstream reaches – creates a unique habitat template which supports a wide range of flora and fauna communities (Vannote, 1980). Traditionally, pressures associated with morphological alteration have been predominantly associated with agriculture, as a result of excess water drainage and livestock grazing practices. These drivers have resulted in the primary pressures of increased sedimentation and channelisation. This has resulted in a reduction of complexity, dynamism and biodiversity within the river ecosystem (Elosegi and Sabater, 2013). Channelisation typically increases stream bank erosion, meaning that more sediment enters and clogs up the stream. This increased sedimentation makes it difficult for some fish to feed and spawn, while the increased velocity of the stream drives out fish which are unable to tolerate fast-moving water. In addition to this, channelisation also reduces the amount of vegetation along the stream bank, which means less food and cover for wildlife. Loss of vegetation further alters erosion dynamics, often accelerating erosion processes. This contributes to highly simplified and uniform channels, unnaturally steep banks and little dynamic connectivity with their flood plains (Brooker, 1985).

As per the characterisation within the WFD, the impact of such morphological alteration will be presented and discussed according to its three constituents: (a) Structure and substrate of the riverbed; (b) River depth and width variation; (c) The riparian zone.

a. *Structure and substrate of the riverbed*

Changes in sediment loading to a river network causes the river to adjust and change channel form, directly impacting instream biodiversity. Loss of habitat complexity can be detrimental to all instream communities but especially fish species which require diversity in habitat conditions. For example, brown trout (*Salmo trutta*) spawn in gravel-bed riffles or runs, young fry congregate in shallow areas and adults prefer deep pools and wood accumulations. The absence of key habitats can, however, lead to a reduction in fish populations, creating a knock-on effect throughout the food chain, as predatory fish species have a significant effect on invertebrate food chains (Katano et al., 2006).

The broader impacts associated with fine sediments on riverine habitats, primary producers, macroinvertebrates and fisheries are presented in a review by Wood and Armitage (1997). In brief, siltation in the water column increases turbidity, limits light penetration to the streambed and reduces primary productivity, with resultant impacts on the rest of the food chain. On the bed of streams and rivers, siltation can modify the substrate by altering its surface conditions and reducing instream habitat complexity. Blocher et al. (2020) found that all fine sediment has profound negative effects on sediment-sensitive macroinvertebrate species, which increased when combined with flow velocity. In extreme cases, fine sediments smother the entire riverbed, changing channel morphology and reducing the availability of suitable habitats for benthic species – plants and macroinvertebrates. In addition to this, siltation can provide sinks for P and other contaminants which also impact negatively upon food chain dynamics (Wood and Armitage, 1997).

Sediment sources arise from poor land management, construction, bankside erosion, runoff from land adjacent to the stream channel and soil compaction which causes poor soil health and grass sward cover (Russell et al., 2001, Gruszowski et al., 2003, Rice et al., 2021, Abbas et al., 2023). Historically, agriculture management practices including drainage, lack of buffers such as catch crops on arable crops, farm roadways, grazing and livestock poaching have been significant sources of sedimentation (Heaney et al., 2001). For example, sediment-fingerprinting research indicated 61% of the sediment load of the River Tweed in Scotland was derived from arable and pasture topsoil (Owens et al., 2000). Specifically, inappropriate grazing and heavy machinery results in widespread soil compaction, which in turn, reduces rainfall infiltration and leads to accelerated overland flow across the catchment. Such changes in the flux of sediment between the terrestrial and river environments in turn directly affects habitat, flow regimes and biodiversity (Wood and Armitage, 1997). This can be intensified due to subsurface field drainage - an extensive practice across NI (Burton, 1986) – as well as channel straightening.

Headwaters draining catchments rich in peat soils are particularly vulnerable to enhanced sedimentation caused by land management. The draining of peatlands can change the hydromorphological condition through the release of fine-grained suspended sediments (Brown et al., 2019). A review by Donahue et al. (2022) examined the effects of physicochemical stressors on instream communities of peatland streams and found there were significant negative changes in aquatic community structure. In Meenboy, Co Donegal, Morton et al. (2024) also found that sediment deposition from eroding peatlands altered headwater macroinvertebrate biodiversity. However, overall, many of the ecological consequences of peat deposition are poorly understood. Organic sediments from areas draining peatland require further examination, to include longer observation periods across trophic levels, with specific reference to habitat conditions across the island of Ireland. This would inform understanding of the extent to which impacts extend from streams into the wider river network.

Contemporary land use can affect sediment processes in rivers further down the river network, especially in heavily modified catchments draining agricultural areas. For example, Sherriff et al. (2015) examined soil erosion and suspended sediment dynamics for Irish catchments and revealed that soil drainage class, together with the proportion of arable land, were key controlling variables determining sediment flux rates. Well-drained soils were less sensitive to erosion even on arable land. It is important to note, however, that under extreme rainfall conditions all bare soils present a high risk of sediment loss.

b. River depth and width variation

A key pressure on river depth and width variation is river management strategies which affect river form. In particular, the straightening, widening or deepening of stream channels as a result of management practices including river channelisation, bank stabilisation and clearing, can have a significant and prolonged impact on biodiversity. River channelisation programmes were undertaken to enhance the conveyance capacity of the river network, primarily to offset the flooding of rural and urban areas but also, to enhance the productive value of agricultural land (see section 4.ii and 5.ii). However, straightening can promote the erosion of both bed and bank materials that were previously in equilibrium during high discharges, potentially leading to bank collapse (Brooker, 1985). In such instances, this causes radical changes to the way a river functions affecting the shape of the channel as well as patterns of erosion and sediment deposition. For example, the enlargement of channel cross-sections can result in a reduced sediment flux through the fluvial system and more deposition of fine substrate within the channel. As an exemplar, channelisation of the River Main in NI led to sevenfold increases in peak sediment loads, as well as increased erosion impairing habitat quality (Wilcock and Essery, 1991). Conversely, sediment deposition can lead to substantial vegetation growth instream requiring repeated maintenance so that channels maintain their design efficiency. This commonly involves removing features, including siltation and instream growth of vegetation within the channel cross-section, via a process called dredging (Bruton and Convery, 1982). This in turn leads to increased erosion of the stream bed even after dredging is completed, causing downstream sedimentation.

Rivers managed in this way have been shown to support fewer species at lower densities than adjacent natural reaches (Brookes, 1985). For example, channelisation destroys the shallow riffles, gravel bars and instream cover which many fish species require for successful spawning. This results in a smooth bedrock that is an unsuitable habitat for macroinvertebrate fauna and fish. In the River Moy, benthic macroinvertebrates decreased by 90% following channelisation (McCarthy, 1981). For the River Main, channelisation destroyed gravel habitat which contributed to a decline in salmon (*Salmo salar*) and brown trout (*Salmo trutta*) redds (Wilcock and Essery, 1991). Furthermore, channelisation has also been shown to disproportionally impact sensitive threatened and endangered fish species, leading to low-quality fish communities and reduced fishing opportunities. For example, in the River Boyne, Ireland, the ratio of salmon and trout to other less valuable species was 14:1 before channelisation. Erosion of and deposition of silty sediments degraded salmon and trout habitats, dropping this ratio to 5:1 following channelisation (McCarthy, 1981). Brookes (1985) stated that the rate of recovery for fish populations from the effects of channelisation has been shown to be extremely slow, some streams showing no significant recovery after 30–40 years. Through increased erosion risk, channelisation is also recognised as being a major threat to sensitive species such as freshwater pearl mussels (*Margaritifera margaritifera*; Österling et al., 2010, Horton et al., 2015). More broadly, species requiring instream vegetation for nesting and/or feeding, such as the moorhen (*Gallinula chloropus*) and sedge warbler (*Acrocephalus schoenobaenus*), are more vulnerable to the direct effects of management, due to disturbance and loss of habitat (Campbell, 1988).

c. *The riparian zone*

Riparian corridors are home to a diverse range of species resulting from their characteristic variability in flood regimes, geomorphic channel processes and vegetation cover (Naiman and Decamps, 1997). This dynamic environment therefore supports a variety of instream life history strategies as species adapt to the disturbance regimes and habitat conditions along the river network (Nakamura et al., 2000, Riley et al 2019). Riparian management, particularly associated with agriculture (e.g. drainage, poor grazing practices) and urbanisation, have simplified the physical structure of riparian habitats, severely modifying their natural flow and ecosystem function. For agricultural catchments, land improvement measures via arterially drained channels have modified the riparian zone. Such practices typically result in a narrow corridor along the margins of the bank's full channel. These stream banks are typically covered by short grass rather than tree structure, a radically different state to the structural diversity of the natural system which increases the risk of erosion

Livestock management practices can also impact riverine biodiversity. For example, livestock access to streams for drinking purposes changes the bank profile and damages, or removes, protective vegetation cover. This in turn can increase riverbank erosion and sediment loss (Evans et al., 2006, O'Callaghan et al., 2018, O'Callaghan et al., 2019). Further investigations in the Irish context also demonstrate that, where cattle are permitted access to streams for drinking, has also led to significant impact on instream communities. However, these responses can demonstrate a high degree of variability, according to site characteristics and seasonal changes, as observed for macroinvertebrate communities (O'Callaghan et al., 2019, O'Sullivan et al., 2019, O'Neill et al., 2023). Moreover, the study by O'Sullivan (2019) for Irish agriculture catchments demonstrated that cattle access has a greater impact on sites with at least GES, emphasising that impact varies according to environmental condition.

Loss of bankside vegetation cover due to management practices also reduces crucial shade which acts as a refuge for macroinvertebrate and fish communities (Broadmeadow et al., 2011). A study in lowland rivers in Ireland found that physical habitat modification corresponded with eco-hydromorphological state (the degree of ecological, riparian and physical modification) and a reduction in thermal buffering capacity. This directly impacted the fish community, with a shift from brown trout (*Salmo trutta*) dominated fish assemblage to a predominance of the minnow (*Phoxinus phoxinus*) and stone loach (*Barbatula barbatula*) which are more thermally plastic (O'Briain, 2019).

Management practice which causes degradation of habitat conditions for native species also typically creates opportunities for invasive species. This has resulted in riparian corridors being subjected to direct physical damage from invasive non-native plant species, which then negatively impact sedimentation and flow. For example, invasive aquatic plants such as the floating pennyworth (*Hydrocotyle ranunculoides*) and Parrots feather (*Myriophyllum aquaticum*) alter habitat conditions by encouraging slower flows and trapping sediment. Moreover, common invasive plants which inhabit riverbanks include Japanese knotweed (*Reynoutria japonica* syn. *Fallopia japonica*), Himalayan balsam (*Impatiens glandulifera*) and Giant hogweed (*Heracleum mantegazzianum*). These species can also contribute to sedimentation as winter die-back exposes the banks, making them more susceptible to erosion (Stockan and Fielding, 2013).

II. Key pressures impacting river continuity

The construction of physical barriers and arterial drainage, together with urbanisation, impacts river continuity over the longitudinal and lateral dimensions of catchment connectivity respectively. Urbanisation and drainage works increase connectivity in the lateral dimension of river continuity, increasing the speed of water movement within a catchment. In contrast, physical barriers disruption the flow dynamics and physical properties, reducing connectivity along the longitudinal dimension. Barriers such as dams, weirs, sluices, culverts, fords and ramps also influence the natural sediment transportation, resulting in the retention of sediment upstream and the loss of sediment downstream. This can cause deepening of the riverbed downstream and a change in the suspended sediment budget. It can also alter the exchange of water and sediment among adjacent habitats within the river corridor as well as between the terrestrial-aquatic environment (Quinlan et al., 2014, Kelly-Quinn et al., 2022).

The key impact of these drivers on river continuity is habitat fragmentation. This impacts upon habitat diversity and, in turn, affects the movement of fish species within the river network as they migrate upstream and downstream. Indeed, their cues for seasonal movements, which are provided by water flow and organic matter, are subsequently influenced by the overall condition of the river (Franklin et al., 1995, Bunn and Arthington, 2002). For example, barriers can impact upon the migratory pathways of species such as Atlantic salmon (*Salmo salar*), sea trout (*Salmo trutta trutta*), sea lamprey (*Petromyzon marinus*), river lamprey (*Lampetra fluviati*lis) and European eel (*Anguilla Anguilla*) by preventing or reducing their ability to move within the river network. Although the impact of barriers on diadromous fish is well-established, the impacts on river-resident fish communities remain unclear. Barrier impacts may also be significant for potamodromous species (e.g. brown trout (*Salmo trutta*) and pike (*Esox lucius*) on Lough Beg, whose entire life cycle is completed within rivers but who are known to make extended movements for feeding or to find spawning grounds (Birnie-Gauvin et al., 2017, Birnie-Gauvin et al., 2018).

Other key issues include habitat isolation through fragmentation causing populations to completely separate. This results in failed recruitment of new organisms to communities and ultimately, leads to local extinction (Wilkes et al., 2019). For example, in instances where fish are faced with a barrier, they are likely to suffer as a result of expending high levels of energy to overcome it. Meanwhile they may also sustain injuries or even succumb to death during their attempts. Encountering a barrier can also result in fish swimming further to find a suitable habitat, spawning in unsuitable habitats or not spawning at all. Thus, barriers are widely attributed to the decline in fish populations by negatively impacting upon the free passage between nursery, recruitment, feeding and breeding habitats. However, under certain circumstances barriers can be positive in protecting biodiversity as documented for native European crayfish (such as *Astacus astacus* (Linnaeus, 1758)) by Frings et al. (2013), which has suffered severe reduction since the invasion of the non-native signal crayfish (*Pacifastacus leniusculus*) into European ecosystems (Holdich et al., 2009). They highlighted that physical barriers to crayfish are an effective method to protect indigenous crayfish in streams with sufficiently high flow velocities.

In contrast to physical barriers, land management practices that contribute to soil compaction and removal of excess water, as well as urbanisation, typically increase connectivity over the lateral dimension by modifying flow patterns and increasing sediment in streams (Yang and Toor, 2018, Jani et al., 2020). When areas are developed, permeable surfaces like soil and vegetation are replaced with is hard, sealed, and impermeable ones such as concrete. This drastically reduces the ability of the land to absorb water, meaning rainwater no longer infiltrates into the ground. Instead, it rapidly runs off into storm drains and directly into streams. This results in changes to the natural processes of the stream network and its continuity by burying, straightening, removing or blocking river channels. Ultimately, this reduces the natural storage capacity of the stream network – its ability to slow down the flow and absorb water through floodplains and riparian buffers. This intensifies the speed at which water moves towards the SSN, increasing peak flows during storms. Critical to biodiversity, this can alter the natural flows required for aquatic organisms to survive, leading to loss of habitat and water quality degradation.

Similarly, arterial and land drainage alters the speed and typically increases the natural movement of water through catchments and river network with significant impact on river ecosystems. This is because arterial drainage was historically supported by field drainage operations to increase their effectiveness in removing surplus water from fields for agricultural purposes (Bruton and Convery, 1982). Such drainage operations were conducted at a local scale by individuals to drain their lands. While these effects were localised, evidence suggests that their accumulative significance could be great for water quality. Studies have found that such drainage has contributed to downstream flashiness, as exemplified by the River Main (Essery and Wilcock, 1990, Wilcock and Essery, 1991). However, the degree of response is likely to vary due to the study sites' characteristics, differential drainage patterns and locations within the catchment, alongside differences in study seasons or durations, or variations in climate patterns and antecedent conditions (Robinson, 1990). In Ireland, the freshwater pearl mussel (a critically endangered species) has been particularly affected by the changes to riverbeds and water quality resulting from arterial drainage. This is because freshwater pearl mussel requires stable, clean riverbeds, which are disrupted by increased siltation and altered flow patterns (Beasley et al., 1998, Reid et al., 2012). In addition to the drainage of agricultural land, peatlands were commonly subjected to drainage, which impacted upon their associated stream systems. Holden et al. (2006) indicate that the long-term response of peatlands to drainage differs from short-term responses, thus highlighting the importance of temporal aspect of drainage and potential instream impacts. To conclude, the legacy of arterial and land drainage practices continue to pose challenges – not fully quantified as yet, to instream biodiversity in many NI rivers.

III. Key pressures impacting the hydrological regime

Variation in flows and water levels is important for rivers to maintain their characteristic ecological diversity. For example, higher flows provide a trigger for migratory fish such as salmon (*Salmo salar*) to make their runs upstream and successfully navigate waterfalls and other obstacles to migration (Kennedy et al., 2013, Gardner et al., 2016). Higher flows also play a critical role in sediment transport (Sheriff et al., 2018). This creates the diversity of shifting habitats on which different flora and fauna depend. Modification of natural hydrologic processes disrupts the dynamic balance between the movement of water and sediment that exists in free-flowing rivers. However, the ecological effects of flow regime change are not caused by the driver of hydrological alteration per se (e.g.

water abstraction, diversions or discharges, channelisation, drainage, dams, weirs and agricultural intensification), but rather, by the way in which these drivers alter specific hydrological attributes (Poff et al., 1997, Bunn and Arthington, 2002). For example, flow regime change is evidenced in terms of altered baseflow(s), reduced flooding magnitude and frequency, as well as altering flow variability. A comprehensive review by Poff and Zimmerman (2009) reported that almost all published research found negative ecological changes in response to different types of flow alteration. These collective impacts arose from a variety of direct and indirect causes such as habitat loss and fragmentation, altered water quality and thermal regimes. Additional causes included the loss of important life-history cues, changes to food webs and patterns of energy production, as well as how the modification of environments promote ecological invasion (Poff et al., 1997, Bunn and Arthington, 2002). Poff and Zimmerman (2009) only reported positive associations in ecological metrics in relation to shifts in ecological organisation for non-native species, which are known to degrade the overall health of aquatic ecosystem Maguire et al., 2011). Thus, in summary, this highlights the severity of the threat of altered flows to the ecological integrity and biodiversity of river systems.

River flows are also impacted by urbanisation which disturbs the natural water cycle, moving precipitation rapidly from the point at which it lands, via drainage infrastructure, to the nearest water course. This is referred to as urban runoff (URF), while urban stormwater runoff (USF) is the water that flows through the lined or piped drainage system to receiving waters. Stormwater run-off contains pollutants, including sediment and nutrients, altering flows and impacting the ecology of the river bodies into which it is discharged (LeFevre et al., 2015). As the surface water system is designed to move water rapidly away, it can impact hydrological processes, alongside increasing channel erosion and enlargement by creating unnatural river morphology and habitat.

Research shows that the diversity of algal, invertebrate and fish communities can be adversely affected due to degraded water quality and unnatural flow regimes when the area of impervious surfaces approaches 10% of the catchment area (Paul and Meyer, 2001). This can impact downstream communities, where flooding can occur as a result of altered natural drainage patterns, often compounded by pipped drainage in urban areas. Recognising the threat of urbanisation to in-stream biodiversity, and in particular macroinvertebrate communities, a study by Walsh (2004) on catchment imperviousness and drainage connection – the proportion of impervious area directly connected to streams by stormwater pipes – was undertaken. The outcome of this studies indicates that sensitive riverine taxa were absent from sites with greater than 20% drainage connection. Moreover, this suggests that it is the connectivity of impervious surfaces to the stream network rather than the impervious surface area itself which is most important in terms of instream impact. It is therefore important that urban development considers low impact design approaches that appropriately manage URF and stormwater drainage to reduce the significant threat the present to instream biodiversity.

To conclude, the hydrological regime ultimately underpins river function, so to protect biodiversity the natural flow variability must be maintained. However, despite the central importance of flow to the river ecosystem and its vulnerability to pressures impacting all aspects of the hydromorphology element, there has been limited research on characterising flow-ecology relationships and its impact on biodiversity. At present, the ecological significance of individual flow events and the sequences of events on aquatic communities – and how these linkages may change under future climate forecasts for NI – remains largely unknown. Flow should, however, be given consideration in the context of the natural typology of river catchments and the impact of climate change on the naturalised or uninfluenced flow (Horne et al., 2022). In NI, some methods have been developed, however, to assess future river flows (Kay et al., 2021). Similarly, several studies in Ireland have also evaluated possible future changes in rivers albeit using different study designs (Charlton et al., 2006, Steele-Dunne et al., 2008, Meresa et al., 2022, Murphy et al., 2023). While such assessments are critical to the preservation of habitat, often the biological effectiveness of e-flow regulations has been omitted from evaluations of flow. A comprehensive literature review by Webster et al. (2017) concluded that there was an insufficient evidence base on which to propose bespoke e-flow standards in an Irish context. This work also specified the need for an analysis of flow augmentation-driven ecological impacts in rivers. Thus, there is a critical need to address this evidence gap to better understand the response of biota and riverine ecosystems to flow modification and restoration, including interactions with river morphology, sediment and nutrient transport and the broader catchment dynamic.

Summary

1. River hydromorphology, which involves three key habitat criteria—morphology, continuity, and hydrological regime—has been heavily impacted by human activities, often negatively affecting biodiversity. While the physical impacts are well documented, there is a need to better understand the specific ecological impacts of such pressures at appropriate high resolution and fine spatial scales.

2. The key pressures identified are channelisation and sedimentation which impact the morphology of rivers; physical barriers, drainage and urbanisation which impact the continuity of the river network; and modification to the flow regime from physical barriers, flood protection measures (e.g. channelisation and engineering), land use changes (e.g. urbanisation, deforestation, agricultural) and changing weather patterns due to climate change.

3. There is a need to better understand flow modification as a key pressure on rivers. While most assessments have focused on water and ecological quality, less attention has been given to river flow requirements and the impact of disrupted fluvial connectivity.

4. Climate change is expected to intensify pressures on ecosystems by altering precipitation and temperature patterns. This will lead to more frequent droughts, increasing habitat fragmentation, while also affecting stream flow and sediment dynamics. This requires further investigation for biodiversity conservation.

5. While the importance of hydromorphology for supporting diverse fish and macroinvertebrate communities is well known, there is a growing need to better understand its broader influence on other ecological functions. This includes its effects on plant and microbial communities, as well as the hydromorphological conditions that sustain ecological interactions and the overall biodiversity of river ecosystems. Such targeted ecological focused research is critical for preserving biodiversity and for developing effective conservation strategies for NI rivers.

Part II. Nutrient enrichment: Pressures and impacts

Impacts of nutrient enrichment

The state of the plant community (rooted plants and algae) in a river at any one point in time is dictated not only by the basic environmental character of the river but by the recent history of environmental conditions. These key determinants include current velocity, scour, light intensity, temperature, and grazing pressure. Nutrients, including N and P, are found naturally in healthy river systems, supporting plant growth and the broader aquatic food web. Nutrient availability is typically regarded as a secondary 'modifier' of plant community composition and biomass in rivers, with its effects constrained by the primary driving forces. Moreover, the response of plant species to increased nutrient availability is dictated by the form of nutrients available and the species-specific growth response to the nutrient gradient. For example, some plant species are adapted to very low nutrient concentration (Preston and Croft, 2001).

The addition of P to river environments is responsible for stimulating algae and plant growth (Dodds, 2007), altering the competitive balance between algae and higher plant species and impacting upon biodiversity (Mainstone and Parr, 2002). Rivers are highly dynamic environments (Snell et al., 2019), so the state of the algae community can vary on short timescales (Snell et al., 2014), particularly driven by the rapid response of the algal community to changing environmental conditions. Moreover, within the SSN algal growth is controlled by terrestrial nutrient input and therefore responds to localised point or diffuse sources. Indeed, Snell et al. (2019) demonstrated differences in the community composition of the phytobenthos in relation to P inputs in three agricultural catchments in North-west England. Nutrient enrichment due to P can also influence biodiversity through changes in the composition and increased abundance of rooted macrophyte communities, with a reduction in those species which are adapted to lower nutrient conditions. This can contribute to increased siltation (Jones et al, 2011), resulting in poorer substrate conditions for benthic invertebrates and fish species which typically require open sediments with high interstitial dissolved oxygen concentrations.

P inputs can also affect other components of the benthic food web. For example, moss floras (bryophyte group) typical of upland streams can be significantly altered. Within downstream reaches, persistent P enrichment can encourage the growth of filamentous algae species, such as *Cladophora glomerata* (L. Kütz). This is considered a nuisance alga due to its tolerance of high nutrient concentrations and excessive growths or 'blooms' which alter species composition (Whitton, 1970, O'Hare et al., 2010). Such blooms can have a significant impact on river biodiversity as they reduce the light conditions, causing a shift in community balance towards shade-tolerant species and, ultimately, algal dominance. Such changes in the basal food web can, in turn, impact upon higher trophic levels (macroinvertebrates and fish) through changes in food resources (Heaney et al., 2001, Dodds, 2007). Continued excessive concentrations of these nutrients can further deteriorate instream conditions, with algae blooms consuming oxygen from the water column, undermining ecosystem function (Brooks et al., 2016). In addition, some cyanobacteria may produce toxins which are hazardous to life (Mainstone and Parr, 2002, Paerl and Otten, 2013).

Beyond the effects on the plant community, nutrient enrichment can also create changes within the faunal community which are just as fundamental – and sometimes, even more so. Herbivory can suppress the effect of increased plant growth rates on plant biomass, but the macroinvertebrate and fish communities are altered as a result of increased productivity in ways that are damaging to characteristic biodiversity. As nutrient levels increase, grazers dominate at the expense of shredders, and detritivores (such as chironomids) which feed on easily accessible algal detritus can also be favoured. Concomitant shifts in the fish community can occur as certain types of feeding opportunities increase, associated with increased plant and invertebrate biomass. Moreover, this can increase the stress experienced by surviving fish populations due to dietary changes from the altered macroinvertebrate communities (Schreck and Trot, 2016, Thera et al., 2020).

Some eutrophication studies have looked directly at relationships between enrichment and the secondary effects on various fauna. For example, Graham et al. (2009) looked at competition between Atlantic salmon (*Salmo salar*) and brown trout (*Salmo trutta*) in similar natural Irish streams which were selected to exhibit a gradient of P concentrations. They found that salmon dominated at lower nutrient levels, while trout became increasingly dominant above 30ugl^{-1} soluble reactive P (40ugl^{-1} Total P). This effect seemed to be caused by increased primary production, leading to an increase in primary consumption through invertebrate grazers, which in turn, reduced the energy expended by the fish on foraging. The efficient foraging strategy of salmon at lower nutrient levels is made obsolete as food resources increase with increasing nutrient status. This resulted in the socially dominant brown trout out-competing salmon for territory at higher nutrient concentrations.

To conclude, nutrient additions can lead to significant ecological changes, causing a loss of biodiversity across the entire food web. Beyond the environmental impacts, eutrophication also creates major socioeconomic challenges (Pretty et al., 2003), including reduced drinking water quality, increased water treatment costs, higher flood risks, and negative effects on recreation and amenity value.

Drivers of nutrient enrichment

The main drivers of nutrient additions to rivers stem from agricultural and urban development, including agricultural runoff, wastewater discharge and impervious surface runoff. With population growth and agricultural expansion, nutrient pressures are expected to increase. Research from 2001 to 2009 (AFBI-NIEA, 2016), evaluated nutrient loading in freshwater lakes, estuaries, and sea loughs, identifying the key contributors to nutrient pollution in NI's aquatic systems:

- For N, 83% was attributed to lowland agriculture and 9% to WWTWs.
- For P, 45% was attributed to lowland agriculture and 46% to WWTWs.

More recent, and subsequent, substance flow analysis (SFA) as part of a project examining how to 'Re-focus Phosphorus use in the UK Food System' (RePhoKUs; Doody et al., 2020), further characterised nutrient P sources as agriculture (62%), WWTW (23%), STSs (12%), and industry (3%). This demonstrates that over 60% of P losses to NI water bodies originate from agriculture. Moreover, futher work on the project demonstrated a regional

P imbalance between demand and supply - with supply exceeding demand - resulting in a low P efficiency for the overall food system. Consequently, surplus P accumulation in soils and waterbodies is high, presenting a significant eutrophication risk to waterbodies across NI (Rothwell et al., 2020).

To build on this work, further research and applicaion of nutrient source apportionment is needed to help identify and quantify the specific contributions of the sources outlined through the RePhoKUs project. For example, the Source Load Apportionment Model (SLAM) developed by the EPA for broad-scale assessment of point and diffuse nutrient loads across Ireland. This has enabled estimations of the relative contribution of N and P sources to surface waters across catchments. In the NI context, Adams et al. (2022) applied the SLAM to two catchments – Arney and Blackwater. For Blackwater, the model demonstrated that diffuse agricultural practice (improved grassland) was the largest source of P loss. In total, the diffuse sources comprised 82% of the total P-loading for the Blackwater catchment compared to 89% for the Arney. For the Arney, WWTPs were an important point source load, thus highlighting the importance of catchment scale investigations of diffuse and non-diffuse sources, and source origin (agricultural diffuse sources, domestic septic tanks, WWTPs). This highlights the challenge of diffuse nutrient losses, which originate from specific hotspots rather than uniformly across the landscape, varying by catchment. Nutrient source apportionment helps pinpoint these hotspots, enabling more targeted mitigation efforts. In order to enable policymakers and land managers across NI to implement strategies in a resource efficient manner there is a need for further work on targeting the main sources to eutrophic risk via nutrient source apportionment models.

In conclusion, rivers serve as the main pathways for excess nutrients, particularly to lakes and coastal waters, as seen in Northern Ireland, including Lough Neagh. Nutrient concentrations in rivers result from point sources like sewage treatment plants and farmyards, as well as agricultural runoff and leaching. In agricultural areas, diffuse sources dominate, while urban areas are more affected by point sources. The nutrient release varies based on land use and population density. The following sections will explore these key drivers—land use and wastewater management—and their impacts on river biodiversity.

I. Land use: Forestry and agriculture

Land use within a catchment is a key determinant of P source, with many studies documenting a strong link between land use (agriculture, forestry, peatland) and declining water quality. In NI, only 8.5% of the total land area is afforested (approx. 8% for Ireland). However, with current policy initiatives like the NI Forestry Strategy aiming to convert 12% of land into woodland by 2050 (and consult The Climate Change Act (Northern Ireland) 2022 (Act)), forestry has the potential to put significant pressure on water quality and biodiversity in the future if not managed correctly. For example, the potential negative impacts of afforestation and harvesting processing on water quality are well documented (Clarke et al., 2015, Kelly-Quinn et al., 2016, Duffy et al., 2020, Flynn et al., 2022). In these instances, P losses from forestry arise primarily during planting and clearing, contributing to P enrichment especially in upland catchments (Gibson, 1976). In addition to forest clearing practices, poorly managed and located plantations can also negatively impact upon water quality, contributing to sedimentation and nutrient enrichment of the river network. Through land drainage poorly sited plantations can alter flow regimes. This ultimately contributes to the pollution and modification of waterways impacting the ecological communities present. However, impacts on the ecological assemblage's present can vary. For example, in a blanket peat catchment in Ireland, an assessment of the impact of forest clear-felling on benthic communities found that community diversity declined for macroinvertebrates but had no significant impact on phytobenthic diatom assemblages (O'Driscoll et al., 2013). Furthermore, other studies have indicated long-term positives associated forestry cover over the course of the rotation, thus making it a viable option under land use option in terms of water quality subject to its correct location and management (Duffy et al., 2020). However, further studies are required in terms of NI impacts on instream communities.

Despite the potential negative impacts of forestry, when compared to the extent of agricultural practice, forestry management is a minor contributor to nutrient enrichment of rivers. Agricultural land use dominates NI, occupying 70% of the countryside, and presents a significant pressure on water quality as outlined within WFD River Basin Management Plan for NI (2021-27) and presented in Doody et al., (2020). As a result, seasonal agricultural runoff poses the greatest risk for eutrophication and is commonly considered in terms of the source-pathway-receptor relationship model illustrated via the Phosphorus Transfer Continuum (Haygarth et al., 2005). Given the historical prominence of agriculture across the landscape, there is a long-established relationship between the proportion of agricultural land in a catchment and diffuse P loss (Foy et al., 1995), which is also reflected more broadly within the literature (Crooks et al., 2021). However, the risk of P loss is not uniform across farm enterprises, varying according to what the land is used for – i.e. whether it's grassland or arable land (Regan et al., 2012). Each type of farming will subsequently alter the properties of the soil, with the intensity of modification dependant on the overall management procedures and existing soil condition. Intensively managed grasslands, for example, will have different but potential significantly sediment erosion rates and nutrient losses (Peukert et al., 2014, Peukert et al., 2016).

In addition to soil properties and general management practices, the impacts of land use can also depend on the state of nearby water bodies. For example, Roberts et al. (2016) examined the effects of land use on P transfer to high-status rivers in Ireland between 2001 and 2012 at 508 water quality monitoring sites across the country. This study highlighted the need for better nutrient use efficiency on extensive grassland farms on marginalized land to manage P transfer risk from extensive farms to high status sites. Moreover, agricultural measures for high status catchments should be targeted to field scale, rather than farm scale, with the support of scale appropriate soil geochemical and hydrological data. Overall, this suggests the importance of considering background environment conditions across the terrestrial-aquatic interface in determining the impact of land use on instream conditions.

At an individual farm level, the specific aspects of farming infrastructure (e.g. yards or fields) and management (e.g. livestock, feedstuff, fertiliser or soil management) can result in more direct, and often concentrated, emissions to the stream network. Farm management practices, such as livestock grazing, can negatively impact nutrient loss while stocking density has been shown to impact upon nitrate leaching to rivers. Research from Richards et al. (2015a), however, demonstrated lower losses for low-intensity grazing systems. Similar findings were reported by Melland et al. (2012) in terms of nitrate-N concentrations and farming intensity, which varied according to the farming calendar (e.g. ploughing, seeding) and season. With respect to farm infrastructure, farmyards have long been recognised as an important risk factor for nutrient enrichment via direct emissions to the water course, with subsequent implications for stream health. For example, particulate P is a significant source of P in NI catchments (Douglas et al. 2007), and often originates from farmyards. Moreover, adverse changes in water quality have been strongly linked to direct inputs from farmyards, with high dissolved oxygen and ammonium-N strongly negatively correlated with macroinvertebrate scores (Hooda et al., 2000). Thus, farmyards can be a significant source of P in water bodies, with detrimental direct and indirect effects on instream communities.

Field infrastructure within a farm enterprise and the application of inorganic and organic fertilisers can build up in the system over time (Sharpley et al., 2013, Cassidy et al., 2017), presenting significant eutrophication risk. While farmyards present a P source, the distribution of P across the field infrastructure and its management can present another significant risk to eutrophication in terms of P surplus. Environmental surpluses of P are attributed to agricultural run-off, nitrogen-based fertilisers and animal manure. While a focus on P use efficiencies within the agricultural sector has delivered some improvements in water quality (Heaney et al., 2001, Barry and Foy, 2016), P inputs still remain greater than crop and soil requirements (Doody et al., 2020). Therefore, legacy P impacts can be seen at the field scale (McDonald et al., 2019, Doody et al., 2020), with uneven distribution of P sources evident. These issues are further compounded by ongoing farming practices, which tend to be habitual in nature with respect to nutrient management. Behavioural change plays a crucial role in addressing these issues (Blackstock et al., 2010, Mills et al., 2016), as altering established practices – through for example nutrient management planning - can significantly improve nutrient management across farms (Daxii et al., 2019).

Consequently, it is important to consider the risks of current discharges from P-enriched soil and sediment. This should also include the potential of legacy-liable P within sediment to be mobilised, especially given the increased storm potential, as per the future climate forecast (Li et al., 2023). Therefore, managing P supply to rivers across the landscape – illustrated via the Phosphorus Transfer Continuum (Haygarth et al., 2005, Deakin et al., 2016) - together with instream release from sediments, continues to be important for reducing the negative ecological impacts of primary production associated with increased nutrient concentrations (Chetelat et al., 1999, Bowes et al., 2007, Snell et al., 2019, Li et al., 2023). Quantifying this total available pool of P in a catchment system is critical to informing effective decision-making (Macintosh et al., 2018) – and this presents a critical research gap.

The Phosphorus Transfer Continuum (Haygarth et al., 2005, Deakin et al., 2016) provides a framework to illustrate the movement of P across the catchment to the stream network. Moreover, it highlights hydrology as the primary driver of nutrient delivery, showing how nutrients move from sources and hotspots across the field infrastructure to streams. Water mobilises and transports P through the landscape via overland, subsurface flow pathways or detachment, with particulate phosphorus (PP) and soil particles separated via erosion. These pathway factors for rivers have been documented in various research studies – for example, by Heathwaite et al. (2005), Mellander et al. (2012), Deakin et al. (2016) and Roberts et al. (2017). In brief, areas of high hydrological connectivity of source to the river present a significant risk for nutrient loss (P, N) to the river network. The attenuation potential of nutrients varies considerably with hydrological location and type of nutrient (Archbold et al., 2010, Archbold et al., 2016). For instance, nitrate is typically delivered to streams via subsurface pathways (Kröger et al., 2007, Tesoriero et al., 2009) while the majority of P from diffuse sources is driven by storm events and delivered via overland flow (Jordan et al., 2005). However, significant quantities of P may also be delivered via tile drainage (Monaghan et al., 2016, Zimmer et al., 2016) and groundwater pathways (Mellander et al., 2016). Climate change is also expected to alter pathway dominance and the speed of water movement, driven by hydrology and landscape topography, soil characteristics and vegetation cover (Forber et al., 2018). These effects may be exacerbated where high-risk agricultural practices are located in close proximity to watercourses, on steep slopes or on under-drained land, which increases hydrological connectivity and results in nutrients being delivered to rivers more efficiently.

In terms of receptor impact, the advancement of research in relation to real-time *in situ* high-resolution monitoring through catchments has assisted with elucidation of the sources and timings of nutrient transfer to the SSN (Jennings et al., 2022). In particular, the long-term high-resolution monitoring platform facilitated by the Agricultural

Catchments Programme (ACP) in Ireland, AFBI in NI and the Demonstrations Test Catchments (DTC) Programme in England has evidenced the delivery of excess nutrients, primarily from agricultural catchments, to the river system (Macintosh et al., 2011, Jordan and Cassidy, 2011, Mellander et al., 2012). The data generated from these monitoring platforms demonstrates the importance of the winter period of the arable and grassland studied alongside the response of the resident ecological communities especially the phytobenthos (diatom) community (Snell et al., 2019). Such datasets have also been invaluable in demonstrating the importance of considering that both N and P can act together in controlling primary productivity (Dodds and Welch, 2000, Dodds, 2007, Jarvie et al., 2018, Snell et al., 2019). However, there has been extensive research into the processes and impacts of eutrophication and water quality, the timescale and spatial extent of climate change impacts have been less well studied. This is despite their potential to increase eutrophication risk and influence the expression of eutrophication symptoms presently and into the future.

The different components of climate change (e.g. temperature, hydrology) not only affect multiple levels of biological organisation – from instream benthos to fish communities – but may also interact with the many other stressors that rivers are exposed to (O'Briain, 2019). For example, summer droughts will not only lead to elevated temperatures but, through low flows, could contribute to habitat fragmentation. Alternatively, summer drought characterised by extended periods of drying out of the riverbed followed by high rainfall events could also exacerbate the impacts of eutrophication, altering N:P ratios and their temporal dynamics instream. However, there is a lack of evidence relating to simple patterns of co-occurring nutrient and algae response to nutrient delivery. To inform this, there is a need to better understand the sources-pathway-receptor framework of the individual and relative impact P and N loadings to rivers. In particular, the critical source areas for these pollutants and their distribution across NI needs to be further considered, to allow for more targeted and individualised measures for river biodiversity restoration and protection.

II. Wastewater management: Centralised treatment

Wastewater pollution problems are commonly associated with population growth and the concentration of population in urban centres. From a riverine ecology perspective, untreated sewage contains high amounts of bioavailable P which, as research shows, is a limiting nutrient within river ecosystems (Whelan et al., 2022). To manage this eutrophication risk, among others such as heavy metal and pathogen contamination, wastewater management involves the collection and treatment of wastewater from towns, cities, and villages before returning it safely to the riverine system. This is known as centralised wastewater treatment and has become a critical part of modern infrastructure to treat wastewater efficiently and systematically from residential, commercial and industrial sources. This process is regulated through the UWWTD which sets the standard to be met in the collection and treatment of wastewater as well as outlining the monitoring requirement. In brief, the UWWTD sets the minimum requirements for monitoring the performance of treatment plants and receiving waters, with the objective of protecting the environment from the adverse effects of wastewater discharges. This is critical as numerous studies have documented the potential ecosystem impacts of WWTP effluent on nutrient loads to waterbodies and the associated eutrophication risk (Jarvie et al., 2006, Carey and Migliaccio, 2009, Bowes et al., 2010, Mockler et al., 2017, Civan et al., 2018). Reducing the concentration of P and N in wastewater is essential to prevent eutrophication and protect biodiversity. However, while WWTPs can be effective in reducing nutrient load, the potential remains for effluent to significantly impact the chemical and biological characteristics of the river ecosystem. This is because it is impossible for the composition of WWTP effluent to match the composition of the receiving stream or river.

Although outside the scope of this review, it is important to note that, in addition to nutrients, there are various chemicals present in wastewater effluent due to the increasing amounts of chemicals used by industry and in domestic settings (e.g. see WFD 3rd Cycle River Basin Management Plan 2021-2027). In general, the ability of these chemicals to interact with nutrient pressures and their impact on biodiversity is largely unknown. For example, tailored monitoring programmes have demonstrated reduced diversity, or indeed absence, of sensitive species downstream of WWTPs. It has been suggested that the observed reduction in diversity downstream of WWTPs may be due to the occurrence of chemicals present in effluent discharges (Ginebreda et al., 2010, Bunzel et al., 2013, Stalter et al., 2013, Burdon et al., 2016). However, there are also other pressures present which can confound research outcomes which impairs downstream habitat quality, including fine sediments, increased water temperature and modified community composition (Pilière et al., 2014). This area requires further investigation, especially as effluents are more proactively managed for nutrients, N and P. Moreover, it will be important to support the zero-pollution and circular economy ambition of the EU Green Deal.

Discharge emissions from the sewage treatment network is derived from three key sources in NI. In the first instance, WWTPs form the primary source of water utility discharges in NI, with the composition of the effluent influenced by the density of the population served (Posthuma et al., 2008). However, for NI, combined sewer overflows (CSOs) also form a significant part of the sewage treatment network, releasing occasional discharges from sewers carrying both foul sewage and rainfall runoff water (NIWater, 2024). In addition to these discharges, emergency overflows are emitted from sewage pumping stations under emergency conditions.

To fully characterise the effects of WWTP discharges to rivers additional continuous high resolution monitoring over appropriate spatiotemporal scales is required (Boënne et al., 2014). However, in NI the issues of discharge emission is complicated by the fact that over 70% (equivalent to circa 2,800 CSOs) of the public sewer system is 'combined,' meaning that it was designed and constructed to collect both foul sewage and storm water (McKibbin, 2015, NIWater, 2024). CSOs transport all their wastewater to sewage treatment plants, placing unnecessary strain on the capacity of these facilities whilst also increasing water treatment costs. While CSOs are a necessary part of the system to reduce the risk of overloading during extreme weather events, periods of intense rainfall can exceed the capacity of the system, causing out-of-sewer flooding of untreated foul sewage. This results in CSOs discharging untreated waste directly into the stream network.

In instances when large volumes of water enter the combined system during heavy rainfall, it results in storm overflows (often referred to as spills). As of May 2024, there are

2444 operational storm overflows deployed across the public wastewater network at pumping stations and treatment works – thus resulting in NI having more proportionally more storm overflows per level of population than many other parts of the UK. This an important mechanism to manage diluted wastewater and protect properties from flooding caused by wastewater backing up the sewer pipes during heavy rainfall. However, these spills can have significant environmental impact, particularly on water quality and aquatic biodiversity. In NI for a typical wastewater spill during storm conditions it is estimated that it constitutes 1-2% raw sewage (NIWater, 2024). Therefore, due to the design and capacity of the sewage treatment network, rivers still frequently receive contaminated wastewater and particulate waste either directly from storm overflows or via emissions from sewage treatment facilities. Cumulatively, this contributes to widespread low-level but persistent discharge of such effluents, impacting water quality and instream communities. Subsequently, while urban runoff is an important consideration in terms of nutrient enrichment and associated biodiversity impacts, the influence of inadequately untreated sewage effluent on nutrient concentrations can be orders of magnitude greater than the effect of stormwater. This merits specific consideration in the NI context with respect to the protection of riverine biodiversity. Urgent action is required to address this additional nutrient source to rivers.

In addition to these pressures, WWTPs are subject to daily variations in flows and loads, as well as other seasonal variations, such as temperature, tourist populations, or industrial loads. Given current predictions relating to the climate and scale of population growth for NI (Lowe et al., 2018, NISRA, 2024), existing infrastructure, due capacity shortfalls and aging, may be unable to cope with modern demands nor able to comply with regulatory standards (e.g. WFD, NAP, UWWTD) to prevent waters deteriorating. This can be attributed to the historical underfunding of NI treatment facilities where insufficient financial resources have been allocated to maintaining, upgrading and expanding NI wastewater infrastructure.

While NI's wastewater system is performing below the standard required for environment protection (NIWater, 2024), this situation is not unique to NI. For example, the report 'Urban Waste Water Treatment in 2019' publish by the EPA showed how, in Ireland, only 30.4% of the WWTPs were designed to provide secondary treatment with some form of nutrient removal. As a result, more than 50% of the wastewater generated in urban areas is discharged with a level of treatment below EU standards (EPA, 2019). An update published by the EPA in 2023 documented that there had been investment in treatment infrastructure at priority areas which is facilitating improvement in environmental standards. Specifically, six villages that had daily raw sewage discharges were connected to treatment plants over the previous reporting period (EPA, 2023b). However, while progress is being achieved by Uisce Éireann, Ireland, similar to NI, has still not met all its obligations under the UWWTD, more than 30 years after the required to bring provisions into force to comply with its terms across the island of Ireland. Addressing this issue requires increased effort in strategic planning to specifically target these legacy issues in a manner that places greater consideration on how current infrastructure is impacting river biodiversity and threatening the long-term sustainability of water quality.

In conclusion, while investment has been made across wastewater treatment networks to meet the goals of the UWWTD, further investment and concrete management actions are needed to close the compliance gap and protect riverine ecology. Additionally, urban design must place greater emphasis on mitigating nutrient pollution and preventing structural degradation of rivers to improve water quality and biodiversity. Currently, the built environment does not adequately support biodiversity (Goertzen et al., 2022) (Goertzen et al., 2022), and research on this topic is limited. More research is needed to explore ways to enhance ecological function and biodiversity in urban areas across NI.

III. Wastewater management: Decentralized treatment systems

Decentralized wastewater treatment consists of a variety of on-site wastewater treatment systems (OSWWTS) for the collection, treatment, and dispersal of wastewater from individual dwellings, industrial sites, and in some instances clusters of homes or businesses. These commonly include STSs and package treatment plants (PTPs). Like centralised wastewater management approaches, the purpose of OSWWTS is to reduce the concentrations of contaminants, including nutrients, to acceptable levels before the treated waste reaches water sources, supplies or before people encounter it. Discharges from OSWWTS are regulated under the Water (Northern Ireland) Order 1999 which requires those with, or installing, an OSWWTS to obtain a consent to discharge. The

Domestic Consent Public Register is available online (consult: www.daera-ni.gov.uk/DCPR/), with all consents from 1992 available for download. The Northern Ireland Environment Agency (NIEA) has estimated that approximately 140,000 properties are served by OSWWTS (DAERA, 2024). However, a key issue for water quality and biodiversity is the circa 16,000 unconsented OSWWTS (McKibbin, 2015, Cave and McKibbin, 2016), and it is likely that the number of these systems is underestimated (Withers et al. 2014). Many such buildings have been shown to contribute to the nutrient loads in streams from their STS (Withers et al., 2012). As there is no was legal requirement to obtain a consent to discharge for systems installed prior to 1973, when the Water Act (NI) came into effect, many of the unconsented systems are over 40 years old. A critical point is that little is known about the suitability of the locations of the older unconsented systems for local conditions. Moreover, some of these older systems discharged directly into ditches, streams and rivers and (Withers et al., 2012).

In general, the need for OSWWTS has coincided with the rich farming heritage across the island of Ireland including NI, with one-off housing forming the characteristic settlement. Therefore, STSs have a long history of use for the treatment and disposal of domestic wastewater in rural areas where connection to the mains sewage system is isolated, impractical, and expensive. OSWWTSs therefore typically comprise small, scattered point sources at a range of densities across the rural catchments, making their pollution impact difficult to quantify. Despite the perception that OSWWTSs contribute negligible amounts of nutrients relative to more diffuse land-based agricultural sources, accumulatively they present a significant risk to water quality in certain conditions, especially at local or catchment scale (Withers et al. 2014). The density of tanks, and particularly those that are poorly maintained, has been shown to increase the risk of pollution. For example, a study conducted in three rural tributaries of the Blackwater River found that high densities of STSs – particularly when poorly maintained – increased the risk of pollution (Arnscheidt et al., 2007).

Due to environmental heterogeneity, P leaching from OSWWTS is often difficult to quantity. A report in Ireland carried out by Gill et al. (2015) for the EPA on wastewater treatment for single dwellings reviewed the efficiencies of both PTPs and STSs in terms of the performance of existing soakaways under different soil permeability conditions.

Through the instrumentation of soakaway systems, which were more than 20 years old, it was demonstrated that existing soakaway systems in low-permeability soils are likely to be causing shallow lateral flow of effluent into adjacent surface depressions, thus increasing the risk of surface water pollution. In relation to the NI context, Foy at al. (2003) suggested that just under half of the human P loading was retained by STSs used in the late 1990s, with 58% discharging into watercourses. This information becomes increasingly important from an ecological perspective, when N and P are released into sensitive water bodies. The extent of this impact for the biodiversity of rivers may depend on river habitat condition and ecological status of a stream, as defined by the WFD (O'Donoghue et al., 2022).

Due to the historically low intensity land use and low population density in areas of the west coast, comparative to other EU member status, Ireland has retained many high-status sites. However, there has been a continuous and persistent decline in such sites over recent decades. This network of high-status water bodies is generally negatively related to agriculture intensity. The decline in high status sites therefore suggests effects from localised impacts (Irvine and Ni Chuanigh, 2011, White et al., 2014, Trodd et al., 2022) including STSs that have the potential to disproportionately affect the ecological health of rivers where the environmental standards of receiving water bodies are high. Moreover, with declines in high-status sites (Trodd et al., 2022; consult chapter 6), there is now increasing merit for investigating the performance and impact of STSs over appropriate spatiotemporal scales in catchments (Gill et al., 2005). This research is supported by work conducted under the RePhoKUs project on characterising P sources (Doody et al., 2020). This highlighted the need to investigate STS more broadly across NI, especially as there are no routine inspections of septic tanks after a consent to discharge has been issued. This is also evident from an assessment of OSWWTS in the Blackwater catchment, where 35% of the 113 systems surveyed were at high risk of negatively impacting water quality due to poor system conditions, and 73% were classified as medium risk (Cave and McKibbin, 2016).

In Ireland, the EPA conducts inspections of Domestic Waste Water Treatment Systems (DWWTS) and in 2023 it was reported that 45% (532/1,189) of DWWTS failed inspection. Reasons for failures fell into two categories. The first category related to operational issues of de-sludging and maintenance, while the second category related to structural defects (for example illegal discharges to ditches/streams, leaks, ponding and rainwater ingress; EPA 2023). Moreover, the EPA highlighted that a lack of awareness about the operation and maintenance requirements was evident among homeowners. This highlights a need for better advisory campaigns across the island or Ireland, especially as the regulatory body NIEA, with responsibility for enforcement action with respect to discharges in NI, are only permitted to act where a pollution incident is of medium or high severity. In addition to the need for advisory and educational campaigns, further research is required to evaluate the effectiveness of such guidance on management practices, like de-sludging, in improving the effluent quality of OSWWTS, particularly regarding P concentrations in the effluent (May and Woods, 2015).

More broadly, the lack of research on the specific details of effluent management and its impact on effluent quality makes effective regulation challenging. For example, careful consideration is required in terms of temporal variation in effluent quality and the impact of occasional, seasonal and intermittent use on treatment systems. In general, both PTPs and STSs require a steady flow of sewage to operate effectively (O'Keeffe et al., 2015, May and Woods, 2015). Moreover, effluent quality from more standard installations may vary over time. Gill et al. (2009) examined phosphate concentrations in the effluent from a standard, two chamber, STS serving a single household of 6 residents in Ireland over a 14 month period at 2-4 week intervals. The results demonstrated significant variability in PO_4-P concentrations, ranging between 5 mg l^{-1} and 37 mg l^{-1} over the 14-month period. These findings highlight the need for further research into the uncertainty surrounding individual measurements of effluent quality for management and regulatory purposes.

To conclude, there is a need to better quantify and understand the significance of OSWWTS pollutants compared to other sources such as agricultural land use in rural catchments. This is supported by Withers et al. (2014), who recommended that more accurate accounting of the location, performance and degree of failure of STS is required. This would increase the precision of estimates of nutrient emissions and account for future potential risks to the river network. This research is imperative as, with changing weather patterns and summer drought, STSs and PTPs individually or accumulatively, may present a substantial increasing risk to water and ecological quality, especially in headwaters characterised by reduced dilution capacity and higher connectivity to pollution sources present in catchments.

Summary

1. Nutrient enrichment, leading to eutrophication, is a process driven primarily by the accumulation of surplus nutrients, particularly N and P, in river ecosystems.

2. Key drivers of nutrient enrichment include land use, land use changes, and climate change, influencing factors such farming infrastructure (e.g. yards or fields - agricultural runoff) and management (e.g. livestock, feedstuff, fertiliser or soil management) alongside wastewater management.

3. River water quality degradation is driven by both point and diffuse pollution sources, compounded by other pressures like hydromorphology, which tend to complicate the nutrient-biodiversity relationship.

4. Nutrient enrichment in rivers primarily leads to prolific plant growth, significantly degrading habitat conditions by depleting oxygen levels, disrupting nutrient cycling, and altering the trophic structure of aquatic communities, which ultimately leads to reduced biodiversity. Although research on nutrient addition to rivers is extensive, the full impact of eutrophication on biodiversity remains unclear and warrants further investigation.

5. Effective nutrient pollutant source apportionment, especially from agriculture, is essential for understanding impacts and implementing mitigation strategies. However, underinvestment in wastewater infrastructure continues to pose a significant challenge, risking further eutrophication without substantial upgrades.

6. Additionally, climate change may exacerbate nutrient enrichment through increased winter runoff and reducing river flows during summer, altering nutrient cycles, flow regimes and the dilution capacity for point source nutrient emissions. These pressures collectively threaten riverine biodiversity and require targeted management interventions at reach and catchment scale.

Chapter 8: Evidence gaps for prioritisation

There is an increasing volume of biodiversity information, with numerous projects and reports attempting to provide an overview of key knowledge gaps and important research questions critical for the protection and restoration of biodiversity. Examples include the European Biodiversity Partnership: Biodiversa+ project (Biodiversa+, 2024) and the Intergovernmental Platform for Biodiversity and Ecosystem Services (IPBES; IPBES 2024). However, effective mitigation occurs at regional and local scale, with appropriate measures implemented at the right time in the right place. Therefore, consideration of specific local or regional conditions is critical for effective conservation.

Based on the evidence reviewed the top five priority areas where knowledge is currently lacking – and which therefore need to be addressed to enhance biodiversity across NI – are presented below. These are important to take note of, as sustainable water resource management and biological conversation critically require the integration of social, economic and wider environmental issues in order to mitigate biodiversity loss.

1. Understanding the interconnected drivers of biodiversity
2. Identifying cumulative and emerging threats to biodiversity
3. Implementing e-flows as a strategy for biodiversity protection
4. A national framework for assessing freshwater resources
5. Transboundary cooperation: key to protecting biodiversity

Priority Area 1: Understanding the interconnected drivers of biodiversity

Direct drivers of biodiversity change are relatively well characterised in terms of hydromorphology and nutrient enrichment. However, it is important to consider that there are other drivers outside the scope of this review which merit equal consideration – namely land use change and associated habitat degradation, impacts of invasive species, other pollution sources including anthropogenic chemicals such as pharmaceuticals, polyaromatic hydrocarbons and pesticides. Moreover, these drivers are often interrelated and their impacts possibly synergistic, further contributing to a deterioration in ecological integrity and biodiversity loss. However, the nature and severity of the scalar relationships between land use, land use change and ecological responses are difficult to establish because of various factors. These include the high variety of land uses and scales along with the dynamics of the changes that instream communities experience, such as climate and hydrological regime. Nevertheless, the need to further understand land use practices and the impact of changes within this area remains, especially in the context of discussions on a land strategy for NI (DAERA, 2015). Given the significance of nutrient enrichment and its threat to biodiversity, it is important to consider how it may interact with other drivers of change – and extending to changes in weather patterns - as it is actively managed. It is also important to consider the processes underpinning these direct drivers, such as population and economic growth, technological advancement and changing society perceptions, which form indirect drivers of biodiversity loss and impairment. These indirect drivers are important as, ultimately, effective responses to them will come from a widespread change in societal attitudes. A comprehensive understanding of these direct and indirect drivers of biodiversity loss are critical for sustainable and long-term protection of biodiversity.

In addition to the need to understand the broad trends in the different driver categories of biodiversity loss and ecosystem change (agriculture and urbanisation), critical knowledge gaps also remain with respect to the particular detail of individual driver categories. Over the last decade, alongside national water reporting (see section 6), State of Nature reports (State of Nature Partnership, 2023) have presented results on the sectoral contribution of N and P to the pollution of the aquatic environment. This re-iterates the urgent need for better predication of the effects of drivers on the current status quo and future impacts. Moreover, any further studies should combine high-quality models with integrated monitoring datasets to improve understanding on key ecosystem processes, biodiversity status and long-term effects. To date, for example, understanding the effects that different agricultural types and intensities have on river biota has been constrained by the lack of synthesis of agricultural data, terms of land cover and management intensity. Other constraining factors include the associated pressures and ecological effects within the riverine network, as well as relatively smaller rural enterprises, such as fish farms, which may be overlooked due to their size but can have significant local effects (Foy and Rosell, 1991a, Foy and Rosell, 1991b, Doughty and McPhail, 1995).

To conclude, more precise apportionment of sources within and among all sectors, along with a knowledge of their distribution and connectivity to the watercourse is critical for land managers and evidence-based decision making. This will become increasingly important with respect to the emergent threats from climate change and the need to have better knowledge to proactively address emerging aspects of local and national change. In turn, this allow for the more reliable prediction of the impacts of such change on river species and to mitigate their effects, moving the science on from focusing on remediation to preventative action which is important for sustainable biodiverse systems.

Priority Area 2: Identifying cumulative and emerging threats to biodiversity

Rivers face multiple pressures, both natural and anthropogenic, which often occur together to influence local biodiversity. Assessing multiple pressures on river ecosystems and understanding their combined impact on ecological status is challenging, especially at increasing spatio-temporal scales. Consequently, investigation of biological responses to pressures are dominated by studies of single stressors, with some mesocosm studies exploring key co-occurring pressures (Davis et al., 2018, Davis et al., 2019). This is because the identification of responses to multiple pressures is characterised by high uncertainty (Ormerod et al., 2010, Page et al., 2012, Hering et al., 2015) and evidence is still in its infancy in terms of how multiple pressures interact individually and with each other to influence biodiversity through time across NI. Therefore, multiple pressures that are acting concurrently on river biota – and their combined effects – are generally poorly understood (Vinebrooke. et al., 2004, Nõges et al., 2016, Birk et al., 2020). As such, while this report focuses on dominant pressures impacting biodiversity in the first instance, there must be cognisance that individual pressures do not occur in isolation and thus, there is a need to adopt a multi-pressure perspective when investigating the impacts on biodiversity. This is particularly important when assessing ecological condition, as communities are shaped by individual or cumulative environmental impacts, local natural conditions and climate variability which accumulate over time (Brown et al., 2013, Piggott et al., 2015, Jackson et al., 2016, Jackson et al., 2021, Sabater et al., 2021). Moreover, as key pressures are managed, how they interact with other components of the system may change, leading to emergent stressors. This may alter the sensitivity of the biological communities to pressure or convey apparent resistant to recovery of communities (e.g.

traits that convey tolerance). Species may also be resilient to one type of disturbance but sensitive to another and their relationship may change temporally across NI (i.e. seasonality).

In addition to the need for greater focus on the potential for 'emergent pressure' problems, there is also a need to consider emergent drivers and their potential threat to biodiversity. The concept of emerging threats has been described by Reid et al. (2019) and among the most critical are a) invasive species, b) emerging contaminants, and c) climate.

a. Invasive species

Invasive species are non-native organisms which negatively impact the new habitats they establish themselves in, often producing effects which humans also do not like and which they deem harmful. Unintended impacts of invasive species include alterations to habitats and direct competition for food and living space. Invasive species have also given rise to the introduction of pathogens and disease which can have severe environmental and socioeconomic effects (Maguire et al., 2011). Thus, in general, invasive species have profoundly damaging impacts on rivers by creating direct competition with native species (Gallardo et al., 2016). For example, the Asian clam (*Corbicula fluminea*) is a recent invader first discovered in River Barrow and Nore in 2010 which is posing a significant threat to the river spawning beds of Ireland's wild Atlantic salmon (*Salmo salar*). Resembling riverbed gravel, it reproduces without a mate, so can quickly carpet the riverbed, thereby keeping other species out. It is currently localised to areas of the River Nore, Barrow and Shannon but presents a high impact risk to native species (Caffrey et al., 2011, Minchin, 2014, Minchin, 2017). In addition to the biological and habitat impact, it can further affect service provision e.g. drinking water abstraction and power plants. This highlights the risk present by invasive species because of the threat conveyed to biodiversity, human water resources and wider ecosystem services with an estimated economic cost to the NI economy of £46,526,218 (The Rivers Trust, 2024).

Such impacts of invasive species are common phenomenon for freshwater habitats, which are increasing experiencing the effects of colonisation and the spread of multiple and co-occurrent invasive species (IPBES, 2019). Therefore, the control and management of invasive species is a priority for the conservation of biodiversity globally (Pyšek et al., 2020). In particular for NI, there is a need for a greater understanding of biological invasions as a primary driver of change, alongside the identification and control of introduction pathways. In addition to this, invasive species as a driver of change is a key priority area and a crucial transboundary issue (see Priority area 5 below). It is therefore imperative to address this issue, so an effective biosecurity strategy can be put in place for the island of Ireland.

b. Emerging contaminants

Emerging contaminants (ECs), which are environmental, biological and human health hazards, present key a threat to river biodiversity. ECs refer to a broad spectrum of compounds used in modern society due to the enormous consumption of medications, personal care products, antibiotics and hormones for cosmetic and health reasons. Agricultural practices, construction, industry and other activities also contribute significant contaminants to the environment. Furthermore, anthropogenic activities have also caused biological contaminants like viruses and bacteria to contaminate water sources. These substances are known as emerging or re-emerging pathogens (EPs) because of their apparent virulence, which has caused increasing concern (Galindo-Miranda et al., 2019, Rapp-Wright et al., 2023, Hands et al., 2024). Numerous waterborne illnesses are prompted by biological contaminants, including enteric bacteria, mycoplasmas, protozoa and viruses, which are hazardous to human health. The range of diseases is continuously expanding although, critically, it is the resurgence of diseases previously believed to be under control (e.g. *Cryptosporidium* found in water) that is of considerable concern (Glaberman et al., 2002, Pelly et al., 2007, NIEA, 2019).

To support nutrient source apportionment and understand responses, there is a need to address the main sources of emerging contaminants, giving special attention to WWTP and the related processes which determine the fate and transformation of organic ECs, including polar pesticides. The biological impacts of ECs may manifest at species and community level – for example, through changes in the behaviour of predators and prey and in the survival ration and reproduction (Hands et al., 2024). However, the link between biological community and chemical stress is complex and other environmental variables may covary with chemical pollution (e.g. hydrology and nutrient concentration). Understanding how river communities adapt or resist emerging contaminants (ECs) is a crucial area for future research. This knowledge is essential for assessing biodiversity responses to current and future environmental drivers over time.

c. *Climate*

Climate also presents a significant emergent driver, and while climate inferences are generally made within studies, direct observations of climate impacts on rivers are challenging to disentangle from the concurrent changes in other factors that also affect instream communities. For NI there are few examples of 'natural' conditions, with many rivers being subject to historical and contemporary changes in land use and riparian vegetation cover, modifications to channel morphology and damming. This makes it difficult to clearly identify changes to biodiversity which result solely from climate change. Many of the projections regarding biological response to multiple pressures, including climate change, assume that any such changes will be linear or simply additive, where the net effects of various stressors are equal to the sum of their single effects. For example, Jackson et al. (2016) suggest that the nitrification of rivers is an additive effect when combined with climate change. However, this may not always be the case, as interactions between stressors can be nonlinear, leading to unexpected outcomes. Research indicates that synergistic effects can amplify the impacts of individual stressors, resulting in more severe ecological consequences than anticipated (REF). Such complexities highlight the importance of considering the multifaceted interactions among stressors to better predict the overall impact on biodiversity and ecosystem health

Despite the acknowledged need to study the effects of multiple pressures, research examining climate change alongside other stressors remains limited. For example, more floods and high-to-extreme hydrologic events have been predicted to result in an increased loading of nutrients to rivers (Arnell et al., 2015, Ockenden et al., 2016, Mockler et al., 2017). Such predications, if realised, could have significant impact for NI land management, and especially in the context of the NiD. In turn, this may have direct consequences for instream communities and be compounded for streams which no longer have sufficient riparian shading and are thus exposed to higher water temperatures. In such instances, instream communities may face additional additive increases to water temperatures under climate change. Moreover, climate change may confound current efforts to mitigate nutrient enrichment and the rapidly of change in climate conditions may make it difficult for species to adapt (Brook et al., 2008). The interactions and impacts of multiple pressures on riverine ecosystems represent a significant challenge for freshwater science, policy, and conservation efforts aimed at protecting biodiversity, both in Northern Ireland and globally.

Priority Area 3: Implementing e-flows as a strategy for biodiversity protection

Habitat fragmentation, degradation, and loss due to flow modification are driven by human activities such as navigation, flood risk reduction, agriculture, urbanization, and industrial or domestic water use. These changes impact the quantity, timing, and variability of water flows, leading to altered ecological integrity and connectivity between habitats. Additionally, climate change is expected to accelerate flow modification, further disrupting sediment movement and impacting species' life histories and biodiversity (Bunn and Arthington, 2002). To guard biodiversity and ecosystem services, habitat protection and restoration must focus on managing key hydrological attributes (Poff et al., 1997, Bunn and Arthington, 2002). This understanding is also crucial for assessing the ecological impacts of societal water use and for strategic conservation planning (Tickner et al., 2020).

One approach to mitigating the effect of flow regulation through habitat degradation on river ecosystems is the practice of water allocations via e-flows, which have been described by Bunn and Arthington (2002). E-flows aim to reduce the threat of altered hydrological regimes instream, creating water levels which mimic natural hydrological variability and incorporate a range of flows essential to supporting ecosystem processes and functions (Arthington et al., 2018). To effectively manage e-flows in riverine habitats, it is essential to increase the number and length of free-flowing streams and rivers. As well as providing flow, by accounting for the variability of hydrographs, e-flows also allow consideration of lateral and longitudinal connectivity. However, implementing e-flows requires a comprehensive understanding of artificial influences, such as water abstractions, diversions, and discharges, which are currently under-researched in

Northern Ireland. It is also vital to assess the ecological condition of rivers, focusing on biological quality elements that are sensitive to changes in river flows and levels (Webster et al., 2017). The characterisation of e-flows therefore requires an understanding of seasonal flows over the river reach, of which there is also insufficient data. Overall, this lack of evidence is impeding the ability to set bespoke e-flow standards and undermines any habitat-oriented biodiversity conversation strategies.

Despite advancements in e-flow science across the island of Ireland (e.g. Webster et al 2017), implementing e-flows as a tool for water resource and biodiversity management has progressed slowly due to the challenges outlined in obtaining adequate data, accurate accounting of the physical impediments to flow and the need for long term hydrological records relating to flow-ecology relationships. The NI e-flow standards (NI e-flow river type classification standards (DoE (NI), 2015) developed based on the principles set out in the UK Technical Advisory Group (UKTAG) of the WFD and related environmental standards (UKTAG, 2008), highlight this need to for further investigation into the link between hydrology and ecological responses across various landscapes and river types. This would serve as a critical step in addressing habitat degradation through flow modification, which is a persistent threat to riverine biodiversity, both in NI and globally (Dudgeon et al., 2006).

To conclude, successful river protection and restoration depend on robust data and modelling to inform water allocation within a range set by the resilience of the river system (Poff and Zimmerman, 2010). A holistic approach to e-flows is needed, considering longitudinal flow dynamics and lateral connectivity within catchments and floodplains. This is vital for supporting the adaptive responses of instream communities as habitat conditions change. E-flows are essential for integrated strategies to mitigate multiple pressures on river systems (Tickner et al., 2020) and are crucial for future efforts to protect riverine biodiversity.

Priority Area 4: A national framework for assessing freshwater resources

Biodiversity has come into acute focus in recent times, with the declared Climate and Biodiversity Emergency for Ireland by vote of the Dáil Eireann on 9th May 2019 (see report:

Houses of the Oireachtas, 2019) and research showing that NI ranks 12th worst for biodiversity loss out of 240 countries. This places NI one behind Ireland (Natural History Museum and RSPB report, 2020). Despite this, currently in NI there is no framework for managing human or flora and fauna conflicts over water, which represents an important research gap. Research indicates that effective water management should include strategies to balance biodiversity conservation with the demands of agriculture, urban development, and other human activities. The absence of such frameworks can lead to increased tensions and competition over water resources, adversely affecting both river ecosystems and their communities. A recent case for Ireland by O'Donoghue et al. (2022) examined the influence of economic activities (agricultural, land use, residential and industrial activities) on water quality. It emphasised the importance of a multi-pressure framework approach in the management of water quality risk and for appropriate land use planning aimed at restoring and maintaining water quality as required by the WFD.

Indeed, to manage the recovery, maintenance and protection of water quality with respect to nutrients and biodiversity, there is a need for a joined-up approach. Establishing a National Freshwater Assessment Framework in Northern Ireland will significantly enhance biodiversity protection in rivers by providing a structured approach to monitoring and managing freshwater ecosystems. This should subsequently target agricultural conservation and urban development infrastructure, together with broader stream restoration strategies on nutrient concentration investigated through dedicated assessment programmes. Such a framework will be important for sustainable river management and the fulfilment of national and European targets on biodiversity (e.g. NI Biodiversity Strategy, EU Biodiversity Strategy, 2030). Guidance could be sought from initiatives, such as the Group on Earth Observations Biodiversity Observation Network (GEO BON) Essential Biodiversity Variables (EBVs) and specifically the Freshwater BON and taking into account the objectives 1-5 outlined below.

To conclude, by facilitating data-driven decision-making and fostering collaboration among stakeholders, such a framework can address the complex challenges facing riverine biodiversity, ultimately leading to more biodiverse and resilient aquatic ecosystems

Objective 1: Preserve and expand taxonomic knowledge in river ecosystems

Monitoring in rivers has been taxonomically biased and requires efforts to address existing blind spots. To determine the resilience of riverine systems in terms of biodiversity, more focused monitoring efforts are needed for the lesser-known or monitored organism groups, instream habitats and waterbody types, as well as how they interact. This is because a significant proportion of biodiversity occurs within biological groups and communities of less conspicuous organisms, for which there is incomplete knowledge. For these groups and freshwater communities in general, the challenge lies in developing the expertise to identify and describe the organisms present. For example, macroinvertebrates have been intensively studied in Ireland (Kelly-Quinn et al., 2020) but the diversity within groups at species level is largely unknown due to cost, time and expertise. The importance of traditional taxonomy, together with newer and more novel techniques, such as DNA and imaging-based methods, needs to be incorporated into future biodiversity assessment to progress understanding of the resilience capacity of the riverine network against current and future change. Such work is also important to inform national databases such as the National Biodiversity Data Centre and the National Biodiversity Indicators, which provide data on biodiversity for the island of Ireland.

Objective 2: Defining biodiversity: key metrics and indicators for ecosystem assessment

To support biodiversity assessments, research is needed to identify metrics of biodiversity that can be used as indicators of change and to assess regulatory compliance. Any such metrics must be applicable at multiple scales (from local to national (NI) and across the island of Ireland) and should also include indicators of negative condition (for example invasive non-native species). Currently, there are no established measures or specific monitoring networks for riverine biodiversity in Northern Ireland. Understanding biodiversity in river habitats largely relies on national monitoring frameworks mandated by legislation such as the WFD. While the WFD provides an overview of structural and functional biological responses to human activities, it lacks a comprehensive understanding of biodiversity and ecosystem function. The information collected under the WFD can indicate the condition of rivers and their surrounding landscapes, aiding in diagnosing causes of biodiversity degradation. However, this data may fall short in estimating ecological risks related to land use changes or guiding the selection of alternative strategies to mitigate biodiversity loss.

Objective 3: Scaling biodiversity research for better data integration and conservation

Biodiversity datasets are crucial for understanding and conserving ecosystems, but in NI, they are currently limited in scope and coordination. Existing datasets, such as those on the Atlantic salmon (*Salmo salar*) in the River Bush (AFBI, 2024), provide valuable ecological insights but lack broader application across NI rivers. This makes it challenging to draw robust conclusions about biodiversity pressures and trends.

To address this, an integrated system should link biodiversity data with environmental and land use information at appropriate spatiotemporal resolutions. It should also examine the transferability and scalar application of findings across catchment. This would improve the contextual value of datasets and enhance biodiversity conservation efforts. Furthermore, any framework developed should build upon existing national programmes, deliver multiple outputs, and address critical evidence gaps for maximum efficiency. For example, it could incorporate existing efforts such as the International Union for Conservation of Nature (IUCN) Red List Index and the Connectivity Status Index for rivers (Grill et al., 2019).

Additionally, aligning with initiatives such as the European Biodiversity Partnership (Biodiversa+) would improve monitoring, providing insights into biodiversity status and trends while enhancing conservation strategies (Biodiversa+, 2024). Such integration of data and frameworks is essential for more comprehensive and effective biodiversity conservation in Northern Ireland. To date, no such specific framework exists to adequately address the actions required to reverse riverine biodiversity loss (Harrison et al., 2018) and support global (e.g. Convention on Biological Diversity, Global Biodiversity Framework) and EU (e.g. the European Biodiversity Strategy for 2030) targets.

Objective 4: High resolution monitoring for long-term biodiversity conservation

Long-term high-resolution multi-taxa analysis of stream biodiversity at appropriate temporal and life history time steps are sporadic, which limits our understanding of the consequences of key drivers (e.g. agricultural, urbanisation, wastewater management) of change in the riverine system. The wider long term trophic perspective on biodiversity needs to be addressed also, as organism groups may exhibit differences in sensitivity to nutrients and hydromorphology through time. For example, macroinvertebrates can show both positive and negative local diversity trends over time (Poff and Zimmerman, 2010, Floury et al., 2018).

Long-term monitoring is crucial for accurate modelling and scenario-building, as most biodiversity impact studies currently rely on short-term data (less than five years). Short monitoring periods can make interpreting trends uncertain and open to question, as the length of record can influence the magnitude of trends. Therefore, longer datasets often provide more reliable insights of the effects of environmental drivers on specific species and ecological communities. Additionally, the relationship between biodiversity and environmental pressures is often complex and bi-directional, further emphasizing the need for sustained monitoring to ensure effective biodiversity protection and conservation.

Objective 5: New data sources for conservation through public engagement

Future biodiversity monitoring and assessments should consider the role of novel research methods and data sources to address data gaps on the distribution of riverine species. Monitoring should also be explored in terms of non-traditional data sources, including citizen science and social media – as demonstrated through iEcology, using interest and social media data to track biodiversity (Jarić et al., 2020). Similarly, the Big Windermere Survey project (Freshwater Biological Association, 2024), aims to provide seasonal scientific evidence of water quality within the lake and its catchment, helping to identify priority areas where action can be taken to improve the condition of Windermere – England's largest natural lake. Critically, it involves collaboration across stakeholders and is dependent upon on citizen participation for the collection of data to contribute to the scientific understanding of water quality at Windermere. Notably, this highlights that the greatest potential for further improvement may occur where different methods of data collection are combined. Moreover, for example, using citizen science-based large-scale sampling with molecular detection tools have been shown to be useful in analysing the distribution of great crested newts (*Triturus cristatus*) in the UK, a protected species listed in Annexes II and IV of the EC HAD (Biggs et al., 2015).

Priority Area 5: Transboundary cooperation: key to protecting biodiversity

Biodiversity is affected by drivers of land use (agriculture, urbanisation, forestry) and land use change, instream habitat degradation, invasive species and climate change, which are all transboundary in nature. For catchments across NI such as the Blackwater, Derg and Finn, actions such as nutrient pollution and habitat modification in one jurisdiction can have consequences for biodiversity in another. For example, the construction of a dam in one jurisdiction could reduce river downstream flow in another jurisdiction and impact the migration of fish species. Therefore, regardless of political boundaries there is a recognised need for continued integration and collaborative solutions on the issue of water quality and biodiversity across the island of Ireland. This should build on successful exemplars of cooperation and governance for biodiversity across both NI and the ROI, as demonstrated through cross-border collaborative projects. For example, the INTERREG programmes, Source to Tap (consult: sourcetotap.eu) and CatchmentCARE (consult: catchmentcare.eu), which sought to improve freshwater quality in cross-border river catchments.

Being an island and incorporating two jurisdictions, the island of Ireland faces a considerable challenge in addressing biodiversity loss. This is exacerbated for river biodiversity due to its lotic nature and interdependence with the surrounding landscape. Essential to the protection of biodiversity are inclusive strategies and thinking facilitated by fora across jurisdictions. One such example is the All-Ireland Climate and Biodiversity Network (AICBRN: consult aicbrn.net). This offers a platform to compare biodiversity initiatives across the island of Ireland, facilitating collaboration and research to inform policy and conservation efforts. This collaboration is essential for enhancing water quality and biodiversity by improving our understanding of the relationships between biodiversity dynamics and their pressures. Moreover, such networks foster collaborative efforts and engagement on transboundary issues, which is crucial for addressing shared challenges in biodiversity and climate resilience across both NI and the ROI. Ultimately, such cooperative approaches enhance the effectiveness of conservation strategies and promotes sustainable management of natural resources throughout the region.

Public participation in the governance of transboundary water resources is another critical component of achieving biodiversity and water quality targets. Public participation is broadly understood as involving catchment stakeholders, including communities, non-governmental organisations, and private businesses, in the management and decision-making process. The benefits of public participation include wider societal awareness, innovative thinking, and ongoing collaboration with all sectors of society. Critically, it improves the empowerment of individual stakeholders – for example, governance initiatives such as the All-Ireland Pollinator Plan (National Biodiversity Data Centre, 2021), which brings different sectors together across the island of Ireland to support pollinators. This has demonstrated the benefit of collaborative working across the island of Ireland. Similar initiatives should be welcomed for freshwater systems which enable greater efficiency and effectiveness of policies and their legitimacy.

Chapter 9: Recommendations

To protect biodiversity and support human wellbeing, the following recommendations outline the key changes needed to ensure robust riverine strategies which focus on the unique ecology of rivers and the multiple threats which they face.

I. Rethinking river biodiversity: beyond traditional metrics

The protection of riverine life is critical, given their ecosystem services, diversity and intrinsic value, as well as the multifarious pressures and levels of threat they encounter presently and will encounter in the future. Biodiversity is a broad concept relating to the genetic, species and ecosystem diversity and it extends to the variability seen within and between species and their habitats. Biodiversity itself is commonly evidenced by the number and distribution of species and their risk of extinction. However, solely focusing on species, such as those under Red List protection, does not capture the full extent of biodiversity change. This brings several challenges when measuring biodiversity response to the drivers and pressures present in the riverine environment, which is heterogeneous and dynamic in nature. Given the constantly changing environment, interconnectedness to the surrounding catchment and sensitivity to land use change, there is a need to review the concept of riverine biodiversity with respect to research scope, biodiversity definition and metrics employed to assess it. The long-term sustainability of the river network requires that biodiversity measures reflect spatio-temporal heterogeneity and functional process alongside species and habitat diversity.

II. River habitat and biodiversity: the critical role of river sources

For riverine biodiversity restoration and conservation, there is a need to protect a wide range of different freshwater habitat types, including headwater streams, riparian zones, ponds and other small wetland habitats. This should also include the separate designation of HMWBs under the WFD, which differ substantially from river habitats typically examined within the scientific literature. These overlooked habitats deliver critical ecosystem services and support a substantial proportion of extant species and community diversity. Specifically for the river network – and in terms of the scope of this review – the predominance of small streams in the landscape, their biodiversity significance and their susceptibility to anthropogenic pressures due to connectivity with the adjacent land, are well recognised (McGarrigle, 2014, Biggs et al., 2017, Riley et al., 2018). However, such streams are not routinely assessed, nor are they monitored under the WFD, which requires river water bodies to have a catchment area greater than 10 km^2 (European Commission, 2003). Research has therefore tended to focus on key biological groups of downstream reaches as defined under the WFD. Consequently, streams which are the least monitored river habitat, may be an underutilised resource for improving water quality and biodiversity (Meyer et al., 2007, Biggs et al., 2017, Bieroza et al., 2024). This habitat therefore requires greater attention to determine its role in combating trends in ecological status under the WFD and biodiversity, as for example explored by Baattrup-Pedersen et al. (2018) for Denmark.

III. Navigating change: functional diversity and species rearrangements

Naturally functioning river environments determine their local communities by filtering the species present using intertwined hierarchical factors acting on multiple spatial and temporal scales (Poff, 1997). In addition to this, riverine systems have been exposed to pressures which can be grouped into four main categories: catchment disturbance (e.g. land use), pollution (e.g. nutrients, hydromorphology), water resource development (e.g. flow alteration) and biotic factors (e.g. community composition, invasive species) which threaten water quality and alter natural biodiversity patterns. At reach scale in rivers, communities' responses to anthropogenic pressure can manifest in compositional shifts, such as rare species being replaced by more common species, without impacting the overall species richness metrics. To adequately evaluate biotic responses to change, there is a need to give greater consideration of how anthropogenic factors cause not only biodiversity loss but also biodiversity rearrangements. Moreover, community aspects such as temporal turnover, dominance, evenness (species abundance distribution) and succession processes should be given greater attention, as such information can be missed in the routine assessment of biodiversity and ecological health. This would provide a more comprehensive measure of ecosystem function, not just 'special' habitats and species, emphasising the importance of the biodiversity of all organism groups for resilience and adaption to change. Re-establishing functional biodiversity could therefore serve as a focus for river conservation initiatives which allow for habitat heterogeneity and corresponding increases in species diversity.

IV. Enhancing public engagement: flagship habitats and species

From a personal viewpoint, people are more likely to value biodiversity if they understand what it is and why it is important – which is highly dependent upon their direct experiences with nature. Therefore, the urgency of river biodiversity conservation is greatly undermined by its apparent invisibility to much of society, promoting an 'out of sight, out of mind' mentality that limits public engagement and thus, concern. To increase engagement with riverine biodiversity loss, visually appealing habitats and charismatic species should therefore increasingly be used to act as ambassadors of riverine biodiversity (Kalinkat et al., 2017). This could also extend to equally fascinating but typically more hidden species, such as algae, an organism which has featured in community projects across the UK such as the 'Slime watch project' (Creekside Discovery Centre, 2023). Flagship species are also well recognised as being valuable engagement tools used by many European freshwater management and conservation organisations when interacting with members of the public. For example, flagship species have been used within initiatives in the EU LIFE programme (French: L'Instrument Financier pour I'Environmement), the EUs funding instrument for the environment and climate action (CINEA, 2024).

More broadly, public participation and knowledge transfer forms an essential part of supporting land managers and assessing the potential risk of eutrophication to watercourses for safeguarding biodiversity. For example, The Nitrates App (now called the Deltares Aquality App) in the Netherlands (Rozemeijer, 2018) and the River Water Quality App (Qualité Rivière) in France (Les Agences de L'eau, 2019). Biodiversity strategies in NI should therefore consider the conservation efficacy of flagship habitat and species, supported by wider tools for land managers, to promote public, political, and wider stakeholder engagement by bringing rivers ecosystems 'to life' and motivating public and cross-sectoral conservation action.

Chapter 10: Conclusion

This review has sought to enhance understanding of river biodiversity and the pressures which are negatively impacting upon its health – primarily, nutrient enrichment and hydomporphology. As such, it has therefore addressed each of these issues in detail, looking at how N and P (in particular) affects river biodiversity, along with modification to hydromorphological elements such as physical barriers and arterial drainage. It has identified key drivers of these issues as being urbanisation and agriculture, with various studies used to evidence this, although more research is needed in key areas.

Structural changes in the agricultural sector, together with the intensification of farming practices, has resulted in a long-term decline in water quality because of nutrient enrichment. Increasing urbanisation is further contributing to nutrient enrichment and overall degradation of the river habitat. The consequences are manifold, from increased eutrophication potential to highly modified channels, flow regimes and riparian zones. This has resulted in accelerated and detrimental loss of biodiversity, increased opportunity for the prevalence of non-native species and altered ecosystem function.

Positive conservation action has contributed to sustained benefits across scales, from local to national, via a variety of mechanisms, including regulatory instruments (e.g. WFD), fiscal incentives (e.g. agri-environmental schemes) and social actions (e.g. restoration activities by the Rivers Trust). However, the processes by which anthropogenic nutrient enrichment results in adverse effects on the biological communities of rivers are highly complex. They include disruption to the competitive balance between plant species as well as the effects on the fauna which depend on the plant community for food, shelter and reproduction. To protect humans, biodiversity and nature as a whole, there is a need to manage river ecosystems collectively across all sectors, recognising their value as a resource for humans as well as for nature and its biodiversity. This review has identified a clear need to close the gap between how the river environment is currently managed and how future management should look. This should take into account the present condition of the riverine network and address the recommendations and evidence gaps presented within this review. There is also a need of greater incorporation of existing and future evidence into policy, allowing it to inform ongoing practice and strategies.

To date, recommendations to address the immediate threat to biodiversity and the underlying drivers of biodiversity impairment and loss have predominantly focused on specific conservation strategies under the HaD and WFD's 'Programme of Measures' for river systems. Moreover, while there has been increasing ambition for biodiversity action (e.g. the EU Green Deal and EU Biodiversity Strategy, 2030), existing national monitoring does not sufficiently cover all biodiversity facets nor their environmental drivers. Meaningful engagement with these targets and legislative ambition requires a dedicated and systematic programme to monitor changes in riverine biodiversity and to guide policy responses commensurate with the urgency of the current biodiversity crisis. These should be made against the critical areas for action highlighted within this review, and in particular:

- To improve water quality in terms of nutrient addition through better nutrient source apportionment and investment in wastewater management.
- Accelerate the implementation of environmental flows including the restoration of connectivity with priority given to the removal of barriers.
- Examine the contribution of river habitats and their diversity, especially those which have been largely neglected, such as headwaters, to the wider freshwater system.
- Improve ecological knowledge on all facets of biodiversity through the provision of a National Freshwater Assessment Network, to allow for better accounting of biodiversity within and among river communities.

To conclude, beyond the legislative requirements, it is important to appreciate that biodiversity – and in the context of this review, specifically riverine biodiversity – is of vital importance to our society. Supporting an abundance of organisms and sustaining both plant and animal life, the riverine ecosystem is also a key part of human society and deserves our protection both now and into the future. Whether it's providing drinking water or serving as a resource we can use in agriculture and for many more purposes, it plays a pivotal but sometimes, underappreciated, role in our lives. This review therefore calls for a recognition of the importance of our rivers, together with the broader freshwater system in NI, and demands that action be taken to ensure their biodiversity and general health going forward.

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Appendix

Method Statement

To ensure that the inferences drawn within this literature review were not biased by an uneven research effort, a systematic review was undertaken according to the standard method developed by the Collaboration for Environmental Evidence (CEE; www.environmentalevidence.org), which is the leading authority on this methodology in environmental sciences. The purpose of this review was to present key evidence on the impacts of hydro-morphological changes and nutrient enrichment on riverine biodiversity for NI. The systematic review also aimed to identify key evidence gaps and provide recommendations for further research.

Development of search strings

The systemic review was performed to investigate two specific questions regarding the drivers of biodiversity in riverine habitats in NI:

> **Q1:** What is the impact of hydromorphological change on NI riverine biodiversity.
>
> **Q2:** What is the impact of nutrient enrichment on NI riverine biodiversity.

Consequently, two separate search strings were developed relating to hydromorphology and nutrient enrichment as drivers of biodiversity loss. Here, and within this review, NI refers to NI only, Ireland refers to the Republic of Ireland (ROI), and the island of Ireland refers to both NI and ROI.

The different dimensions of biodiversity for benthic algae, macroinvertebrate, macrophyte and fish along with the key terms describing the drivers of nutrient enrichment and hydromorphology were explored as per **Table 1.**

Table 1: Terms and component parts of key words considered for search strings

Terms	Key words describing the term
Biodiversity	**Biota:** Fish, macroinvertebrates, macrophytes, algae/diatoms/phytobenthos **Measures of biodiversity:** species, community, ecosystem, richness, evenness, rarity. **Ecosystem:** function, structure
Nutrient enrichment	nutrient enrichment, eutrophication, pollution, diffuse pollution, nitrogen, phosphorus
Hydro-morphology	sediment, channelisation, flow, discharge, aerial drainage, dredging, impoundment, barriers, dams, weirs, connectivity
Rivers	Rivers, streams, headwaters, small stream network, low order, beck
Location	Ireland, Northern Ireland, UK, England

Partial search strings for drivers

From the key words identified, partial search strings were developed to capture each aspect of the drivers of biodiversity in river habitats (**Table 2**).

Table 2: Partial search strings for search engines for impacts of drivers on biodiversity

Biodiversity	**Biota:** Fish, OR macroinvertebrates, OR macrophytes, OR algae/diatoms **Measures of biodiversity:** communit*, OR assemblage*, OR taxon, OR species*, OR composition richness, OR biotic interaction*
Eutrophication	“nutrient enrichment”, OR eutrophic, OR eutrophic*, OR nutrient rich*, OR pollut*, OR nitrogen, OR phosphor*, OR nutrient*, OR nit*, OR nitrate*, OR diffuse poll*
Hydro-morphology	“sediment, OR sedim* OR channelisation, OR flow, OR discharge, OR “aerial drainage”, OR dredging, OR impoundment, OR barriers, OR dams, OR weirs, OR connectivity
Rivers	stream* OR river*, OR headwater*, OR “low order”, OR beck
Location	Ireland, Northern Ireland, UK, England

Each partial search string was combined using the Boolean operator AND to identify studies assessing the impacts of the specified drivers on biodiversity. These search strings were used in Web of Science and Google Scholar to find published studies based on their titles, authors, keywords, and abstracts. **Table 3** provides different examples of full search strings that were used to identify studies examining the impacts of different drivers of biodiversity change.

Table 3: Example of full search strings for impacts of drivers on biodiversity

Search Strings
Biodiversity* (Topic), and eutrophic* (Topic), and stream* OR river*, OR headwater*, OR "low order", OR beck (Topic) / Ireland
communit* or assemblage* or taxon species* composition richness or biotic interaction* (Topic), and "nutrient enrichment", OR eutrophic*, OR "nutrient pollut*", OR nitrogen, OR phosphor*, OR nutrient*, OR nit*, OR nitrate*, OR "diffuse poll*" (Topic), and stream* OR river*, OR headwater*, OR "low order", OR beck (Topic) AND Ireland
Algae OR diatoms (Topic), and "nutrient enrichment", OR eutrophic*, OR "nutrient pollut*", OR nitrogen, OR phosphor*, OR nutrient*, OR nit*, OR nitrate*, OR "diffuse poll*" (Topic), and stream* OR river*, OR headwater*, OR "low order", OR beck (Topic) / Ireland
Algae OR diatoms (Topic), and "nutrient enrichment", OR eutrophic*, OR "nutrient pollut*", OR nitrogen, OR phosphor*, OR nutrient*, OR nit*, OR nitrate*, OR "diffuse poll*" (Topic), and stream* OR river*, OR headwater*, OR "low order", OR beck (Topic) / Ireland
Macroinverte* (Topic), and "nutrient enrichment", OR eutrophic*, OR "nutrient pollut*", OR nitrogen, OR phosphor*, OR nutrient*, OR nit*, OR nitrate*, OR "diffuse poll*" (Topic), and stream* OR river*, OR headwater*, OR "low order", OR beck (Topic) / Ireland
fish* (Topic), and "nutrient enrichment", OR eutrophic*, OR "nutrient pollut*", OR nitrogen, OR phosphor*, OR nutrient*, OR nit*, OR nitrate*, OR "diffuse poll*" (Topic), and stream* OR river*, OR headwater*, OR "low order", OR beck (Topic) / Ireland
Macrophyte* (Topic), and "nutrient enrichment", OR eutrophic*, OR "nutrient pollut*", OR nitrogen, OR phosphor*, OR nutrient*, OR nit*, OR nitrate*, OR "diffuse poll*" (Topic), and stream* OR river*, OR headwater*, OR "low order", OR beck (Topic) / Ireland
Biodiversit* (Topic), and sediment, OR sedim* OR channelisation, OR flow, OR discharge, OR "aerial drainage", OR dredging, OR impoundment, OR barriers, OR dams, OR weirs, OR connectivity (Topic), and stream* OR river*, OR headwater*, OR "low order", OR beck (Topic) / Ireland
communit* or assemblage* or taxon species* composition richness or biotic interaction*(Topic), and sediment, OR sedim* OR channelisation, OR flow, OR discharge, OR "aerial drainage", OR dredging, OR impoundment, OR barriers, OR dams, OR weirs, OR connectivity (Topic), and stream* OR river*, OR headwater*, OR "low order", OR beck (Topic) and Ireland
Algae OR diatom* (Topic), and sediment, OR sedim* OR channelisation, OR flow, OR discharge, OR "aerial drainage", OR dredging, OR impoundment, OR barriers, OR dams, OR weirs, OR

connectivity (Topic), and stream* OR river*, OR headwater*, OR "low order", OR beck (Topic) / Ireland
Macroinverte* (Topic), and "sediment, OR sedim* OR channelisation, OR flow, OR discharge, OR "aerial drainage", OR dredging, OR impoundment, OR barriers, OR dams, OR weirs, OR connectivity (Topic), and stream* OR river*, OR headwater*, OR "low order", OR beck (Topic) / Ireland
fish* (Topic), and sediment, OR sedim* OR channelisation, OR flow, OR discharge, OR "aerial drainage", OR dredging, OR impoundment, OR barriers, OR dams, OR weirs, OR connectivity (Topic), and stream* OR river*, OR headwater*, OR "low order", OR beck (Topic) / Ireland

Inclusion of databases and other sources

Scientific studies and evidence not captured by the search strings developed in **Table 2** and **Table 3** but considered as potentially important were also considered. To achieve this a manual review of the grey literature was undertaken. European and National policies including strategies such as the National Biodiversity Strategy were also included. Source databases directly documenting the impact of drivers on biodiversity were also investigated (e.g. IUCN Red List of Threatened Species, National Biodiversity Data Centre Ireland).

A selection of specialist websites from Ireland and NI were also searched for relevant publications that may not have been identified through the online database searches. These included:

- The Northern Irish Department of Agriculture, Environment and Rural Affairs website: www. dardni.gov.uk
- The Department of Agriculture Food and the Marine website: www.agriculture.gov.ie
- The Northern Irish Department of the Environment website: www.doeni.gov.uk/niea/
- The Agriculture and Food Development Authority website: www.teagasc.ie
- The Department of Housing, Local Government and Heritage website: www.gov.ie
- The Sustainable Water Network (SWAN) website: www.swanireland.ie
- Uisce Eireann, formerly Irish Water, website: www.water.ie
- The Water Forum website: www.thewaterforum.ie
- Teagasc website: www.teagasc.ie
- EPA website: www.epa.ie
- The Water Joint Programming Initiative (JPI) website: www.waterjpi.eu
- The RIAN website: rian.ie
- Inland Fisheries Ireland website: www.fisheries.ie

- The Northern Ireland Assembly website: www.niassembly.gov.uk
- Catchements.ie website: www.catchments.ie
- UK Technical Advisory Group on the Water Framework Directive: www.wfd.co.uk
- Local Authority Waters Programme (LAWPRO) website: www.lawaters.ie
- NIWATER website: www.niwater.com
- National Biodiversity Data Centre Ireland: www.biodiversityireland.ie
- IUCN website: www.iucn.org
- National Parks and Wildlife Service website: www.npws.ie

Screening criteria and inclusion criteria

Rapid evidence screening assessment of the studies resulting from the output of the search strings was conducted by abstract, title, keywords and location. Studies were then retained for further consideration following assessment of the scoring criteria through steps 1-10 as outlined below.

1. **Relevance of population**: appropriate to the NI context.
2. **Exposure**: assessment of the impacts of different driver(s).
3. **The outcome**: assessment of ecological impact or evidencing ecological impact.
4. **Type of data:** reviews and meta-analysis considered.
5. **Type of data collected:** empirical data: field studies > mesocosm experiment.
6. **Type of analysis**: empirical data > modelled studies.
7. **Scalar:** consideration of spatial and temporal aspect of the biodiversity indicator.
8. **Study length:** multi annual, seasonality considered.
9. **Consideration of novelty**: theorical or practical application.
10. **Publication year**: priority given to studies over the preceding 10 years.

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