

1 **Two centuries of decline and partial recovery of fish in rivers of post-industrial northeast**
2 **England: lessons for global rivers in the Anthropocene**

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31 **Abstract**

32 Understanding the historical decline in ecological condition of rivers, their fish stocks, and of their
33 restoration, offers one perspective by which contemporary threats to river fishes in less developed
34 regions may be assessed. We explore the decline and recovery of fish populations in rivers (Tyne,
35 Wear, Tees) of post-industrial northeast England. Before ~1850, Atlantic salmon (*Salmo salar*) and
36 sea trout (anadromous *Salmo trutta*) were abundant in these rivers. These catchments were subject
37 to heavy industry and urbanization, river barrier construction, and industrial pollution between ~1850
38 and the 1960s, during which time their fish stocks dramatically declined. Following industrial decline,
39 closure of mines, and improvements in urban wastewater treatment, salmon and sea trout started to
40 recover in the Tyne and Wear; these are now the most productive recreational salmon fisheries in
41 England. Anadromous salmonid abundance in the Tees remains low, probably largely due to an
42 estuarine barrage built in the 1990s. Fish communities in these rivers have changed over time,
43 through natural responses and species introductions. Complex environmental impacts remain
44 across the catchments. Many localities fail to meet 'good ecological status', with river fragmentation
45 and diffuse pollution remaining key problems. We conclude that responses of river fish to industrial
46 and urban development are long-term and cumulative. Ecological recovery of river fishes, following
47 severe environmental impacts, requires much time and effort and is rarely fully achieved. This
48 emphasizes the need, worldwide, to avoid severe ecological damage of river systems during their
49 development by society.

50
51 **Keywords:** river restoration, water pollution, migratory fish, fishery management, river barrier

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61 **1. Introduction**

62 Worldwide, many rivers have suffered severely from pollution, habitat change and connectivity loss
63 (Belletti et al., 2020; Dudgeon et al., 2006; Meybeck & Helmer, 1989). Although anthropogenic river
64 habitat modification has occurred for millennia in some regions, globally the situation has worsened
65 in recent decades as development pressures intensified (Dudgeon et al., 2006; Thieme et al., 2023),
66 leading to deteriorating ecosystem functioning and declines in native biodiversity, including fish
67 populations (Deinet et al., 2020; IUCN, 2024). However, such impacts have not progressed similarly
68 in all areas of the world. In western Europe, for example, much of the anthropogenic damage to
69 rivers occurred during the Second Agricultural (~1500-1850) and First Industrial (1760-1840 in
70 Britain, throughout the 1800s in continental western Europe) Revolutions, when rivers became
71 highly fragmented, and large amounts of urban and industrial pollution occurred and was
72 discharged to rivers (Hoffmann, 1996; Nützmänn et al., 2011; Whelan et al., 2022; Winiwarter et al.,
73 2016). These changes were further exacerbated during the late 19th to mid- 20th Industrial Age
74 (Langford et al., 2012; Whelan et al., 2022). Together with overexploitation, this resulted in reduced
75 abundance or extirpation of fish species such as Atlantic salmon (*Salmo salar*), sensitive to the
76 altered conditions (Lenders, 2017). Since the late 20th Century much emphasis has been placed on
77 attempting to restore the functionality of these chronically damaged rivers towards their natural state
78 (Roni et al., 2008; Wohl et al., 2015).

79
80 In Asia, South and Central America, and Africa, major anthropogenic impacts on most river
81 ecosystems are relatively recent compared with rivers in Europe, North America and Australasia
82 (Grill et al., 2019; Parvez et al., 2023; Thieme et al., 2020). The rate of anthropogenic river
83 modification on rivers such as the Mekong (Campbell & Barlow, 2020; Sun, Galib, et al., 2022; Ziv et
84 al., 2012) and Amazon (Castello et al., 2013; Jenkins et al., 2025) has accelerated in recent
85 decades and threatens native fishes and the sustainability of associated fisheries, especially for
86 migratory fishes. Given that the impacts of river degradation on fish assemblages and populations
87 may accumulate over many decades, even centuries, and be difficult to reverse, historical ecology
88 studies have value for helping to understand the likely implications of current, widespread damage
89 to river fish communities (Beller et al., 2020; Moore et al., 2024).

90

91 This paper examines the ecological deterioration and recovery of rivers and their fish populations in
92 northeast England, a region that was a focus of the Industrial Revolution in Britain. The rivers Tyne,
93 Wear and Tees share similarities in their physical and biological characteristics, and some of the
94 environmental damage experienced, but also exhibit differences in their recovery trajectories. Our
95 aims were to, (1) summarize the historical decline and recovery of fishes and associated
96 environmental quality indicators in the Tyne, Wear and Tees; (2) identify likely contributory factors
97 that led to the near extirpation of Atlantic salmon and sea trout (*Salmo trutta*) in those rivers; (3)
98 evaluate the recovery status of aquatic habitat and fish biodiversity in the three catchments; (4)
99 consider threats to fishes in rivers undergoing rapid development around the world in light of the
100 historical ecology of northeast English rivers, identifying lessons to be applied.

101

102 **2. Materials and methods**

103 **2.1 Study area**

104 The rivers Tyne, Wear and Tees flow eastwards from the Pennine Hills to the North Sea (Figure 1)
105 and are flashy ('spate') rivers, in which flow responds quickly to local rainfall. The Tyne is 121 km
106 long, the Wear 110 km long and the Tees 137 km long, with basin areas of 2943 km², 1321 km²,
107 and 1930 km² respectively. The upper parts of the rivers run through upland rural environments,
108 dominated by moorland, with low human population density. By contrast the lower reaches,
109 especially the estuaries, flow through the urban areas (Figure 1) of Newcastle Upon Tyne (hereafter
110 referred to as Newcastle) and Gateshead (Tyneside), Sunderland (Wearside), and Stockton and
111 Middlesbrough (Teesside). During the Industrial Revolution these urban areas became heavily
112 industrialized, persisting until the mid-late 20th Century, but most heavy industry has now been lost
113 and the areas can be described as post-industrial (Linsley, 2003).

114



115

116 Figure 1. Locations of the Tyne, Wear and Tees basins, and major urban areas (grey shaded area)

117 in the UK. Reservoirs, the Tyne-Wear-Tees water transfer tunnel and several tributaries referred to

118 in the text are also shown.

119

120 Tyneside and Wearside were major shipbuilding and manufacturing centres during the Industrial
 121 Revolution (Dodds, 2011; Linsley, 2003), while Teesside was dominated by the steel and chemicals
 122 industries (Sheail, 2000; Warren, 2018). Heavy industry in the region was based on coal as an
 123 energy source and this was deep-mined across the lower catchments of the Tyne and Wear
 124 (Linsley, 2003). The North Tyne is impounded by Kielder Reservoir (Figure 1), one of the largest
 125 artificial lakes, by volume, in western Europe (Archer, 2003a), while the upper Tees has multiple

126 reservoirs, originally built to provide water supply for the heavy industry of Teesside (Warren, 2018).
127 A Tyne-Wear-Tees connector pipeline (Figure 1), opened at the same time as Kielder, in 1982, to
128 transfer water from the Tyne to the Wear and Tees (McCulloch, 2006). In terms of biogeography
129 and fish communities, the study rivers exist within a transitional zone since, within Great Britain, the
130 rivers of the south and east of England have greatest fish diversity and more warmwater species,
131 especially cyprinids, than rivers in the north and west (Wheeler, 1969). Hence, although the rivers of
132 northeast England are dominated by diadromous coldwater fish species, including salmonids, they
133 also contain elements of the fish fauna found in the Humber basin, immediately to the south,
134 including some warmwater species, as well as several introduced species (Supplementary
135 Information, Table S1; (Sun et al., 2020; Wheeler, 1969).

136

137 **2.2 Methods**

138 Historical literature and data sources were used to reconstruct trends in abundance and catches of
139 salmon and sea trout in the study rivers and to gather information on the distribution of other fish
140 species. The same approach was used to gather information on patterns of pollution, river
141 modification and environmental restoration, to provide context as to the pressures acting on fish
142 populations and their associated fisheries. Sources used included journal articles, reports from river
143 boards, fishery boards and commissioners, the Ministry of Agriculture, Fisheries and Food Standing
144 Committee on River Pollution (SCORP), Environment Agency (EA; and its forerunner organisations)
145 reports and data sources, as well as books, newspapers and magazines. Many of these documents
146 were obtained from the UK National Archives (<https://www.nationalarchives.gov.uk>), as well as local
147 historical archive organisations. Online sources of historical information were mainly gathered from
148 the British History Online digital library, British Library Newspapers Archive, Google Books and UK
149 Government Web Archive. For fish species, search strings including “salmon”, “trout”, “smelt”,
150 “herring”, “shad”, “eel”, and “lamprey” were used. For human actions, search strings including “fish”,
151 “fishing”, “net”, “netting”, “catch”, “dam”, “weir”, “lock”, “sell”, “sold”, “mine”, “mining”, and “pollution”
152 were used. For study area, the name of each catchment “Tyne”, “Wear”, “Tees”, and major
153 tributaries / localities such as “Derwent”, “Rede”, “Kielder”, “Browney”, “Greatham”, and “Skerne”
154 were searched. Documents originating as far back as possible were examined.

155

156 Annual declared river rod catches of Atlantic salmon (referred to as salmon below, unless otherwise
157 stated) and sea trout from the study rivers since 1867 were obtained from records of the relevant
158 authorities (see Supplementary Information, Methods). Catch per unit of effort (CPUE) data were
159 calculated from 1993 onwards when angler effort (rod days) became available. Long-term trends of
160 commercial salmonid catch data in the vicinity of the study rivers, since 1951, were gathered from
161 'Salmon and Migratory Trout Statistics for England and Wales' reports (Supplementary Information,
162 Methods). Yearly upstream fish counts for adult salmon and sea trout recorded by electronic
163 resistivity fish counters (see (Lucas & Baras, 2000; Van Der Waal, 2014) on the Tyne (since 1996, at
164 Riding Mill weir, 42 km upstream from river mouth), Wear (since 1995, Framwellgate weir, Durham,
165 22 km from river mouth) and Tees (since 2011, at the Tees Barrage, 16 km upstream from river
166 mouth) were gathered from EA records. Although the fish counters monitor only part of the river
167 width at a key passage route for salmonids at an obstacle on a given river, each counter is operated
168 in a consistent manner, calibrated and quality-checked, and so provides trends in abundance
169 (Gregory & Gough, 2002). Quality-assured river and estuary water quality data were obtained from
170 the EA's national database for the duration of records for sites chosen, spread across the
171 catchments (Figure S1)

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173 Fish sampling records for the Tyne, Wear and Tees estuaries were obtained from the EA and its
174 predecessors, the National Rivers Authority (NRA) and Northumbrian Water Authority (NWA). A total
175 of 9,552 records were obtained from 1981 to 2018, 94.1% of which were beam trawl samples which,
176 for consistency, were used for analyses. Sample sites were divided between the upper (weakly
177 brackish), middle (intermediate salinity) and lower (strongly brackish) estuary reaches. Greatly
178 reduced sampling occurred after 2010, so analysis was limited to the period 1981-2010. Non-metric
179 multidimensional scaling (NMDS; (Kruskal & Wish, 1978)) ordination plots were generated to
180 visualise temporal changes (Matthews et al., 2013) in the estuarine fish community composition
181 using the vegan package (Oksanen et al., 2019) with averaged annual data of each estuary from
182 combining middle and lower reach data. For the Tees, the lower and middle reach sites were all
183 downstream of the location of the Tees Barrage and so remained tidal after the construction of the

184 barrage. Fish species richness was plotted against year for separate reaches of each estuary.

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186 Fish community data from non-tidal river reaches were obtained from the EA and its predecessors.

187 Most quantitative data were from single-pass and multiple-pass electric fishing in stop-netted

188 sections, though a few sites were sampled by seine net, principally to capture juvenile cyprinids

189 (see Supplementary Methods for further details). Although NWA undertook freshwater fish sampling

190 in the 1980s, available records dated from 1991.

191

192 **3. Results**

193 **3.1 Historical abundance of salmon, sea trout and other fish in the study rivers**

194 Before, and for most of the Industrial Revolution, Atlantic salmon were abundant in northeast

195 England's rivers, including the Tyne, Wear and Tees. During the Late Middle Ages (1300-1500),

196 groynes and 'fish weirs' were built in the region's rivers to guide salmon into fixed nets and traps or

197 into shallow water where fish could be netted or gaffed (Champion, 2003). Between 1348 and 1437

198 monks, residing on the banks of the lower Wear, sold salmon and trout caught there

199 (Commissioners for the British fisheries, 1861). Local convents bought hundreds of locally-salted

200 salmon annually between 1530 and 1536 (Commissioners for the British fisheries, 1861). Around

201 1760, a condition was inserted into workers' contracts in Newcastle and nearby towns, that required

202 them not be compelled to eat salmon more than two or three times per week (Anonymous, 1861;

203 Bewick, 1862), reflecting the plentiful abundance of salmon. Early 15th Century records detail

204 numerous fisheries and fish weirs for lease on the Tees (Anonymous, 1928). Smelt (*Osmerus*

205 *eperlanus*), abundant in the Tees, was commercially fished in the estuary during the 16th Century

206 (Maitland, 2003). In 1530, the use of 'kydyll' (kettle) nets - large, fixed trap-type nets - was

207 prohibited in the Tees for catching smelt; a smelt fishing season (25 April to 1 August) was

208 introduced; and smelt could only be fished in the upper estuary (Anonymous, 1928; Brewster,

209 1796). In the 18th Century, the principal fishery in the lower Tees was for salmon, with the surplus

210 sold to regional cities (Anonymous, 1928). Flounder (*Platichthys flesus*) and European eel (*Anguilla*

211 *anguilla*) were recorded as abundant in the Tees estuary (Parson, 1827) and non-migratory brown

212 trout (*Salmo trutta*) were abundant throughout the freshwater part of the Tees in the late 18th

213 Century (Brewster, 1796).

214

215 Salmon provided the main fishery in the Tyne until the end of the 19th Century when superseded by
216 marine fisheries. Over 2,400 adult salmon were caught in the Tyne on 12 June 1755, over 2,000 on
217 20 June 1758, and 260 were caught in one net haul on the tidal Tyne on 6 August 1761 (Mackenzie,
218 1827). A large drag net was used to catch salmon between the upper estuary and Dinsdale weir (18
219 km upstream of the tidal limit) on the Tees. The fish weir at Dinsdale, built around 1750, was 2.4 m
220 high with two fish locks (which acted as salmon traps, not fish passes), one on each bank
221 (Supplementary Information, Figure S2), and fish could only ascend the weir when it was drowned
222 by high flows (Commissioners for the British fisheries, 1861). During the spawning season, in the
223 late 1700s, Tees salmon and sea trout were protected from fishing, so they could reach spawning
224 areas further upstream (Brewster, 1796). In the 19th Century, the Tees was still an important salmon
225 river, with around 10,000 fish netted from the river in 1867 (Netboy, 1968). Few records exist for sea
226 trout fishery captures in northeast England until the 19th Century, because sea trout were often not
227 deemed worthy of recording (Champion, 2003). From the 19th Century, the coastal net fishery for
228 salmon developed greatly and large numbers of salmon and sea trout were caught annually
229 (Champion, 2003). These coastal fisheries intercepted salmonids moving along the coast to multiple
230 rivers and so do not necessarily reflect catches allied to nearby rivers (Potter & Swain, 1982).

231

232 **3.2 Deterioration of ecological conditions in the study rivers and their effects on the fish**

233 **3.2.1 Heavy metals**

234 Fish populations of the study rivers have been exposed to pollution for a millennium - lead mining in
235 the North Pennine Hills (especially the South Tyne, upper Wear, and upper Tees) can be traced
236 back to the early Middle Ages (Raistrick & Jennings, 1989). However, large-scale lead and zinc
237 mining in the headwaters of these rivers started in the early 17th Century and by the 18th Century
238 they were Britain's most important lead and zinc mining area (Archer, 2003a; Archer et al., 2003;
239 Raistrick & Jennings, 1989). The upper reaches of these rivers were badly polluted due to mining
240 activity and ore washing (Archer et al., 2003; Commissioners for the British fisheries, 1861; Hudson-
241 Edwards et al., 2008). In the Tyne catchment, most lead and zinc mines were located in the South

242 Tyne and River Derwent (Figure 1) areas, where salmon nearly vanished in 1870 due to lead
243 pollution (Archer, 2003b; Marshall, 1992). Dissolved lead bioaccumulates in vertebrates and is
244 highly neurotoxic to eggs and larvae of fish, including salmonids (Jeziarska et al., 2009; Lee et al.,
245 2019). Most of the tributaries of the study rivers have sufficiently hard waters of neutral to high pH,
246 due to outcrops of limestone, such that heavy metal pollutants do not dissolve readily, limiting the
247 geographic extent of heavy metal impacts on the fish fauna (Kelly, 1988). Nevertheless, mining
248 activities in the study rivers caused much sediment input, likely with resultant impacts on salmonid
249 eggs (Champion, 2003). Metal mining resulted in severe zinc, arsenic, and lead contamination of
250 the upper Wear and has had a long-lasting effect on river sediment (Lord & Morgan, 2003; Warren,
251 2018) The phytotoxic effect of heavy metals on riparian vegetation is a major factor causing bank
252 instability and the slow recovery of many mining-affected river systems (Hudson-Edwards et al.,
253 2008).

254

255 **3.2.2 The coal industry**

256 Coal mining and processing caused widespread pollution in the Tyne and Wear, particularly
257 between 1720 and the 1960s (Archer et al., 2003). Much of the Tees, except for some tributaries of
258 the Skerne (Figure 1), lay outside the main coalfields. Mining in northeast England's coalfields
259 peaked in the early 20th Century, with local employment in the industry increasing from 10,000 in
260 1810 to 223,000 in 1919 (Galloway, 1969). Most of the coal mines were in the middle and east of
261 the region (Galloway, 1969), resulting in the upper reaches of the rivers being less impacted by coal
262 mining. Most of the coal output was from drift mines and deep mines, so associated pollution was
263 mostly from coal washeries and mine dewatering, as well as from nearby coking plants (Archer et
264 al., 2003). Coal washing discharge caused the riverbed to be covered with large amounts of
265 inorganic particulates, filling gravel interstices and smothering the benthic ecosystem, reducing
266 hyporheic flow and causing substrate deoxygenation (Ghose, 1999; Hynes, 1960). Some coal
267 mines, in localities with soft water, caused acid mine drainage (Archer et al., 2003; Hynes, 1960) but
268 the greater problem was from suspended solids (Hynes, 1960), particularly flocculated ferric oxide,
269 precipitated when pH is near neutral or weakly acidic, deposited on the riverbed and destroying the
270 natural benthic community (Vuori, 1995).

271

272 Because northeast English coal occurred in localized seams, 'pit villages' sprang up to work the
273 coal, resulting in numerous sources of coal mine pollution. During the 19th Century, many pit villages
274 developed coke works at which coal was processed to coke and its byproducts (Archer et al., 2003;
275 Galloway, 1969). Coke production generated large amounts of polluting material including polycyclic
276 aromatic hydrocarbons (PAHs), naphthalene, benzenes, phenols, quinolines, tar acids, cyanide and
277 ammonia (Archer et al., 2003; Liu et al., 2017), which were leached or washed into waterways and
278 are toxic to fishes and other biota (Baumann & Harshbarger, 1995; Liu et al., 2017). The Wear was
279 classified as "ruined by mines" in the late 19th Century and fish conservation locally had been given
280 up entirely (Anonymous, 1867; Commissioners for the British fisheries, 1861; House of Commons,
281 1876; Lord & Morgan, 2003). The problem persisted through the early 20th century (Box 1); for
282 example, the middle reach of the Wear carried high levels of toxic coal tar acids in September 1928
283 (Longwell & Roberts, 1929) (Box 1). Although the northeast England coal fields declined from the
284 1950s to the 1980s, coal production there in the mid-20th Century was still 20-30 million tons per
285 year (Galloway, 1969; McParlin, 2011), and the rivers continued to suffer severe pollution from coal
286 mining, coke works effluents and spoil heap runoff in the 1950s (Whitton et al., 1998).

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Jun 1922 – Tyne. Dissolved Oxygen (DO) of Tyne estuary varies from 18% at 10 miles from river mouth to 84%, 3 miles from river mouth, and 62% at tidal limit. DO consistently <50% throughout tidal cycle 10 miles from river mouth. Smell of creosote where River Team enters Tyne (Meek & Jee, 1922).

Jan 1925 – Tees. Pattinson & Stead (1925) document pollution sources in Tees Estuary, including effluent of ~500 gallons per minute from Whitwell & Co., containing naphthalene (11.2 parts per 100,000), tar acids (6.5 pp100,000), tar bases (3.4 pp 100,000), and 300 gallons/min from Eaglescliffe Chemical Co., with pH < 2, containing ferrous sulphate (5,707 pp 100,000) and free sulphuric acid.

Jul 1925 – Tyne. DO survey of Tyne estuary, towards end of ebb tide, 22 July 1925, after 1 month's drought (worst-case oxygen-demanding conditions). Eight miles of estuary with DO content <15%; 3 miles of estuary anoxic - corresponding with the greatest inflow of sewage effluents from densely populated areas of Tyneside and effluent inflows from riverside industry (R. Gill, 1926).

Apr 1926 – Tyne. "Heavy mortalities of salmon smolts" observed in estuary at Newcastle; associated water samples taken contain naphthalene levels up to 2.02 pp 100,000 (R. Gill, 1927). 75,000 gallons of effluent per hour from Teams Byproduct Company works to River Team, close to confluence with Tyne, including coke washings, naphthalene, benzol condensers, tar acids.

1927 – Wear. Middle reach of the River Wear is "a badly polluted area where the rod fisheries have been practically exterminated". Invertebrate fauna around Durham dominated by pollution-tolerant species. Pollution especially linked to effluents from colliery "by-product plants - effluents high in tar, ammonia, naphthalene, benzol, tar acids and the like" (Jee, 1928b).

1928 – Wear. River Browney, near confluence with Wear (middle reaches) - badly polluted (DO 30%, ammonia 1.15, tar acids 2.5 pp100,000) - receives effluents from Sadlers chemical works, Langley Park Colliery, Coke ovens. In winter, dilution of 1pp 100,000 is toxic to trout; < 0.5 pp100,000 toxic in summer (temperature and DO dependent). Tidal Wear DO, 43-62% (Longwell & Roberts, 1929).

1932 – Tyne. Teams Byproduct Co. effluent flow in 1927 was 120,000-130,000 gallons per hour, including toxic effluents from tar-producer plants, naphthalene condenser water and coke washing water. Dozens of major industrial and sewage effluent sources along Tyne estuary, identified 1922-1931. Trout experimentally exposed to River Team water in summer overturned within 2 minutes, while those exposed to Newcastle & Gateshead Gasworks effluent died in 2 hours (H. O. Bull, 1932).

1935 – Tyne. 2,011 dead salmon and sea trout smolts recovered in tidal reaches between 30 April and 17 June. DO highest during April (but frequently 40-60% DO), very low in May (DO 3-51%) and June (0-30%). Cyanide detected on 4 of 23 sample days - up to 0.1g /100,000, especially near mouth of River Team (severe pollution inflow), where greatest mortalities occurred (H. Bull, 1935).

Sep 1936 - Wear. (Roberts & Leach, 1937) state "Reduction in pollution would make the Wear a fairly productive river for salmon and sea trout". Major pollution concern is the amount of sewage released into estuary from Wearside, as well from towns upstream. Water quality survey: River Browney, DO 40%, ammonia 2.8 pp 100,000; River Wear, Durham DO, 52%; estuary DO mainly 45-60% across 10 stations, Ammonia 0.01-0.09. Concludes that Wear is badly polluted; upper section in best condition, and middle section worst – pollution sources "quarries, coal mines, chemical works, lead mines, coke ovens, gas works, sewage plants, crude sewage)".

1940 – Tees. (Grindley & Pentelow, 1940). Untreated sewage discharged into estuary increased by ~10% from 6,381,472 gallons /day in 1925 to ~6,965,725 gallons /day by 1940 (population of ~0.25 million). Estuary DO, 6-41% on 21 May 1940, typical of May in preceding 15 years, due to industrial waste and sewage (Grindley & Pentelow, 1940). Widespread smolt mortality 1925-1940, largely attributed to poisoning by cyanide from cokeworks and by hypoxic estuary conditions. Skerne (major tributary in lower freshwater reaches of Tees; human population in Skerne catchment ~125,000): polluted throughout its length, mostly due to poorly-treated sewage, coal washings and waste from coal by-product sites. Skerne DO ~80-90% in upper reaches but 10-30% further downstream; macroinvertebrate fauna dominated by pollution-tolerant taxa, especially oligochaetes and chironomids.

301 **3.2.3 Industrial and urban pollution of the lower rivers and estuaries**

302 Throughout the Industrial Revolution, and until the 1960s, diadromous fishes, such as salmon,
303 faced intense industrial and urban pollution in the lower river and estuary of the study rivers, where
304 the greatest urban development occurred (Box 1) (Environment Agency, 2008). Along the Tyne
305 estuary, coke, iron, and chemical works became widespread, together with tanneries, other heavy
306 industry and transportation (Linsley, 2003), and by the mid-19th Century these had created large,
307 concentrated areas of pollution (Box 1). The population of Newcastle increased from 27,500 in 1811
308 to 128,500 in 1871 (Archer et al., 2003). Domestic wastewater was released directly into the Tyne
309 estuary and by the mid-19th Century had resulted in severe oxygen depletion, ammonia elevation,
310 and coating of the estuary bed with toxic sediments. Although intense and chronic pollution
311 dramatically impacted fish populations in the lower reaches of the study rivers (Box 1), during wet
312 weather, high river flows diluted pollution sufficiently to make diadromous fish migration possible on
313 some occasions. For example, due to a spate, oxygen levels in the Wear estuary on 8 July 1927
314 were 73-84% (Jee, 1928a), suitable for salmon migration.

315
316 Frequent mortality of salmon smolts due to pollution in the Tyne estuary was reported in evidence to
317 a Royal Commission in 1891. In 1895, the dry spring caused low freshwater flow into the estuary,
318 and poorly diluted pollution killed thousands of salmon smolts there (Marshall, 1992). Smolt
319 mortalities in the Tyne were commonplace throughout the 1920s and 1930s, partly blamed on toxic
320 chemicals such as cyanide, a coking byproduct, but also because of frequent hypoxic conditions
321 during the spring smolt migration (Box 1). In the 1930s, Tyne salmon sold in Newcastle and London
322 fish markets possessed an unacceptable tainted tarry flavour (Champion, 2003; McParlin, 2011). In
323 1959, after years of low catches, there was no reported rod catch of salmon in the Tyne, probably
324 reflecting negligible numbers of salmon (Champion, 1991). In 1960, there were 275 major
325 discharges of untreated wastewater to the Tyne estuary (Archer et al., 2003), but no effective
326 legislation to control this until the Rivers (Prevention of Pollution) Act of 1961 and Control of
327 Pollution Act in 1974. By then, the Tyne salmon population was almost eliminated (Champion,
328 2003).

329

330 Sunderland, on the banks of the Wear estuary, became the world's largest ship building town in the
331 late 18th and early 19th centuries (Short & Tetlow, 2012). The shipbuilding works led to a heavily
332 modified physical estuary environment, as well as industrial pollution. Disposal of untreated sewage
333 from the growing population in Sunderland, 146,000 people by 1901 (Short & Tetlow, 2012), to the
334 Wear estuary caused more pollution.

335

336 Water quality in the lower Tees and estuary was impacted by untreated sewage from rapidly
337 growing urban areas and due to increasing industrial pollution (Box 1), including from coke-oven
338 activities and steel production (Sheail, 2000; Watson & Davis, 1995). Discharges to the Tees in the
339 late 19th and early 20th centuries mainly consisted of untreated sewage, coke-oven effluents and
340 pickle liquor, a strong acidic solution from steel works (Southgate et al., 1932). Approximately 4400
341 lb of tar acids and 1800 lb of cyanide were discharged into the estuary daily in the 1930s (Alexander
342 et al., 1936). Large-scale mortalities of salmonid smolts and adults, as well as other fish species,
343 were recorded throughout the 1920s and 1930s in the Tees estuary (Alexander et al., 1936; Brady
344 et al., 1982; Sheail, 2000; Southgate et al., 1932), resulting in the rapid decline of the salmonid
345 fishery (Anonymous, 1938a). It was concluded that this was due largely to cyanide poisoning
346 (Alexander et al., 1936; Anonymous, 1938a; Bassindale et al., 1933; Brady et al., 1982; Sheail,
347 2000; Southgate et al., 1932). By 1935, salmon were virtually extirpated in the Tees, and the
348 invertebrate fauna in the Tees estuary was greatly impacted by pollution (Shillabeer & Tapp, 1989;
349 Tapp et al., 1993). The middle reach of the estuary was almost totally deoxygenated in summer and
350 high concentrations of phenol and cyanides from coke-oven effluents were present (Shillabeer &
351 Tapp, 1989). During the same period the smelt (*Osmerus eperlanus*) population was extirpated from
352 the Tees (Maitland, 2003).

353

354 Further upstream, chemical and biological water quality of the freshwater Tees deteriorated for
355 several kilometres below the confluence with the River Skerne (Anonymous, 1938b, 1938a) which
356 was heavily modified due to industrialization, urbanization, and land drainage (Vivash et al., 1998)
357 (Box 1). From the mid-19th Century to the mid-20th Century the Skerne was heavily polluted by
358 sewage from the town of Darlington and by chemical works discharges; its water was frequently

359 anoxic (Box 1). Fish mortalities due to pollution were widely observed (Commissioners for the British
360 fisheries, 1861) and led to the Skerne becoming fishless by the 1930s (Anonymous, 1938a).
361 Although the Skerne inflow polluted the lower Tees, sufficient dilution from the Tees enabled the
362 non-tidal river downstream to hold substantial numbers of brown trout, grayling (*Thymallus*
363 *thymallus*), dace (*Leuciscus leuciscus*), chub (*Squalius cephalus*), gudgeon (*Gobio gobio*), and
364 roach (*Rutilus rutilus*) (Anonymous, 1938a).

365

366 **3.2.4 Connectivity loss from river barriers**

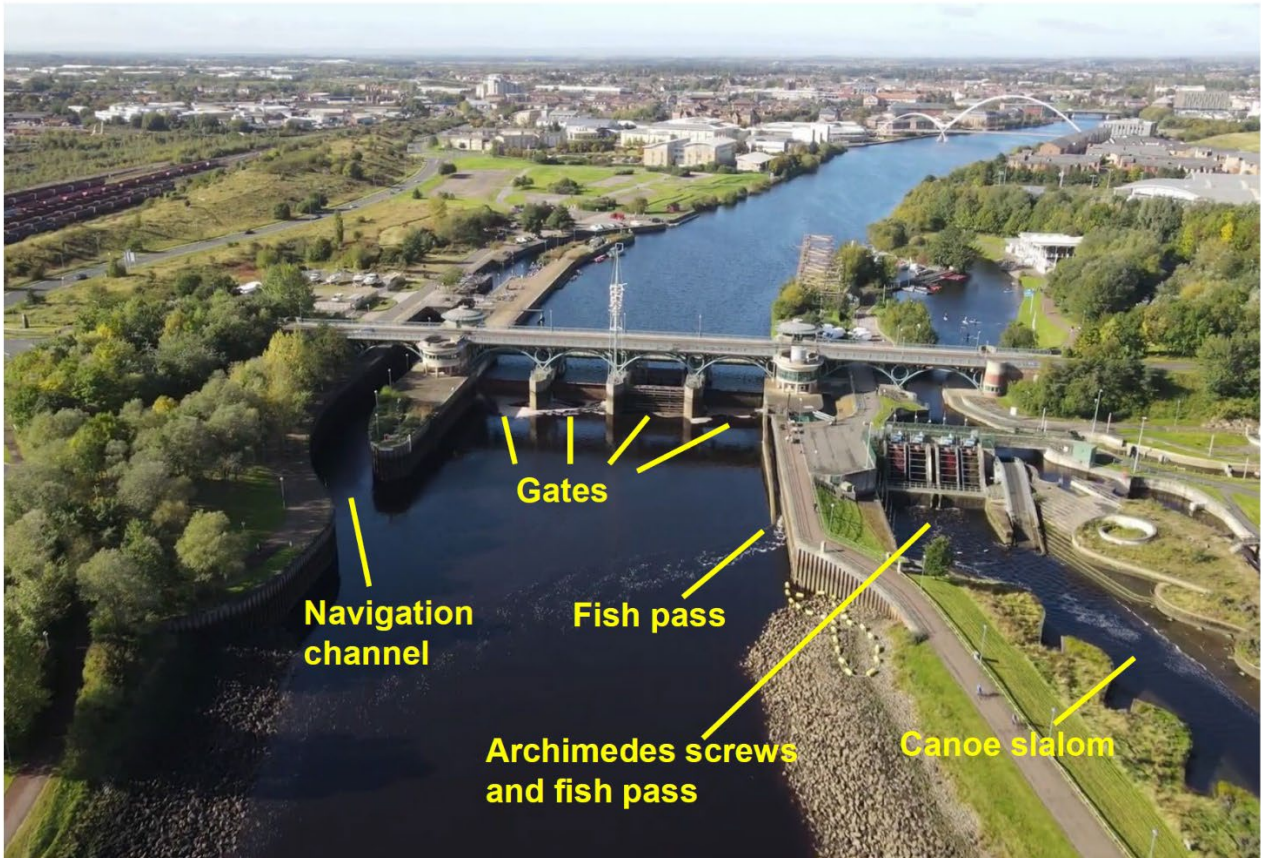
367 Building of numerous weirs in the Tyne, Wear and Tees for fish capture and for milling, from the
368 Middle Ages onwards, undoubtedly impacted longitudinal connectivity for fish migration in these
369 rivers and increased the susceptibility of migratory salmonids to exploitation. However, because
370 these are spate rivers it is likely that substantial numbers of salmon and sea trout were able to
371 migrate upstream when these structures were inundated, particularly during autumn floods prior to
372 spawning (Lucas & Baras, 2001). It is also the case that early wooden and block-stone weirs in
373 northeast England were frequently damaged or destroyed during floods (Anonymous, 1861). Two
374 weirs likely contributing to the decline of Tyne salmon before 1800, were Bywell weir on the lower
375 Tyne, built in the mid-18th Century, and Derwenthaugh damhead, built in 1691 at the tidal limit of the
376 River Derwent, both of which obstructed upstream fish migration, particularly during low flows.
377 Numerous other barriers, including the Warden weir on the South Tyne, and Woodburn weir on the
378 River Rede (Tyne tributary) also impeded fish migration.

379

380 Barriers in the lower reaches of rivers tend to be the most damaging to populations of diadromous
381 fishes because they impede upstream access to most of the catchment (Lucas & Baras, 2001).
382 Salmon and sea trout retained access to the middle and upper reaches of the Wear until the early
383 19th Century, when a dam (Lumley Lock) was constructed at the Wear's tidal limit (Anonymous,
384 1861). Lumley Lock was washed away by a flood in 1820, rebuilt and again destroyed by floods in
385 1854. Multiple low-head dams on the Wear's middle reaches (Anonymous, 1877; Commissioners
386 for the British fisheries, 1861), as well as in tributaries such as the River Browney (Figure 1)
387 (Commissioners for the British fisheries, 1861), impeded upstream migration to spawning habitat.

388 Dinsdale Weir and fish locks (built ~1750, progressively destroyed by floods in the early 20th
389 Century), was the most important obstacle to upstream fish migration on the Tees during the
390 Industrial Revolution. Numerous mill weirs were constructed upstream of Dinsdale weir (e.g.
391 Whorlton lock), most of them installed with fish traps (Commissioners for the British fisheries, 1861).
392

393 Although many of the historic weirs on the study rivers have deteriorated and been destroyed,
394 mostly in floods, new weirs have been built or updated, particularly for river flow-gauging so as to
395 manage flood risk and water abstraction, and some of these barriers have fishways, mostly built
396 between 1970 and 2025 (Table S2; Sun et al., 2020). The single-most important recent barrier
397 impacting fish populations in northeast England, is the Tees barrage, a tidal barrage completed in
398 1994, to manage water flow, limit the upstream tidal spread of pollution and facilitate waterside
399 development (Tapp et al., 1993). After barrage construction, the length of the Tees estuary was
400 reduced to 18 km and the former tidal reach upstream of the barrage became a ponded freshwater
401 reach (Environment Agency, 1999). The barrage has four hydraulically operated gates that, when
402 lowered, allow river flow to spill downstream. There is a navigation lock on the right bank and a
403 canoe slalom facility on the left bank (Figure 2). A Denil fishway (for adult salmonids) and eel elver
404 pass, with their entrances close to the canoe course outflow were installed for upstream fish
405 passage (Watson & Davis, 1995). One or more barrage gates (typically the left-most gate looking
406 downstream) have been used to aid passage of jumping adult salmonids by lowering the gate(s),
407 especially around high tide, but historically, water flow for operating the canoe slalom has been
408 prioritized over fish passage. In 2010, four Archimedes screws were added to the canoe slalom for
409 pumping water upstream for course supply, and a further Denil fishway was added between the
410 Archimedes screws to assist upstream fish passage through the canoe slalom. The two Denil
411 fishways are ill-suited to most of the diadromous fish species of the Tees (Larinier, 2002; Silva et al.,
412 2018). Both are placed suboptimally for fish entry (Larinier, 2002), one has an attraction flow less
413 than recommended [10% of Q_{mean} at Q_{95} ; Larinier, 2002] and the other is accessible for only part of
414 the time.



415

416 Figure 2. The Tees Barrage viewed from downstream.

417

418 3.2.5 Habitat modification from impoundment and gravel extraction

419 Most impoundments in the study rivers are in their headwaters and although these dams have
420 altered river hydrology and habitat, due to their location they have had relatively little impact on fish
421 migration and habitat in the middle and lower river reaches. These reservoirs provide water supply
422 rather than hydroelectric power. In the Tyne catchment, four small (< 100 ha) reservoirs (Whittle
423 Dene, Hallington, Colt Crag, and Catcleugh) were built in the 1800s for water storage and treatment
424 to the Tyneside conurbation (Figure 1). Further demand for water supply by the mid-20th Century led
425 to additional impoundment in the Tyne (Archer, 2003a). Derwent Reservoir (Figure 1), 4 km² in area
426 and opened in 1967, was built on the upper Derwent, to provide water for South Tyneside. Since
427 multiple existing barriers in the Derwent precluded effective upstream fish migration from its lower
428 reaches, access past Derwent dam was not considered an issue, and no fishway was included in its
429 design. Kielder Reservoir (Figure 1), on the North Tyne, was built to meet the predicted demands of
430 industry and domestic water usage in the region and was completed in 1981 (Archer, 2003a). It

431 impounded 11 km of salmonid spawning grounds. Kielder was built legally with no fishway, instead
432 annual stocking of juvenile salmon/eggs was required as a mitigation. A key aspect of the building of
433 Kielder and Derwent reservoirs was to enable their water to be transferred by pipeline across to the
434 Wear and the Tees, for onward (downstream) water supply to Wearside and Teesside respectively
435 (Figure 1).

436

437 The Wear has very little impoundment (three small reservoirs at Burnhope, Tunstall and Waskerley
438 for potable water supply) and, in dry periods, flow abstracted for public supply in the Wear's middle
439 and lower reaches is augmented by water from the Tyne and its reservoirs. By contrast, the upper
440 Tees has multiple impoundments, built to supply Teesside's industrial and domestic needs
441 (McCulloch, 2004). The River Lune has two reservoirs, Selset (opened 1960), and immediately
442 downstream, Grassholme (opened 1915), and then flows 4 km before joining the Tees. The River
443 Balder has a chain of three reservoirs (Balderhead [opened 1965], Blackton [opened 1896], Hury
444 [opened 1894]), and then flows 6 km to reach the Tees (Nelson, 2003). Cow Green Reservoir
445 (opened 1971), contributes 47% of capacity of the Tees impoundments, is the only reservoir on the
446 main River Tees and was planned when water-intensive industries on Teesside were dominant in
447 their influence on water development (McCulloch, 2004). High Force waterfall (10 km downstream
448 of Cow Green) on the Tees has, in recent geological history, prevented all fish passage upstream,
449 while Cauldron Snout, just below Cow Green dam was also a major natural barrier (Crisp, 1984).
450 Therefore, Cow Green has had no effect on fish migrations or the historical decline of anadromous
451 fish populations in the Tees. The Cow Green impoundment has altered local hydrography and
452 thermal conditions (Crisp, 1977) and had local minor effects on populations and ecological
453 characteristics of brown trout, bullhead (*Cottus gobio*) and minnow (*Phoxinus phoxinus*) (Crisp et
454 al., 1984, 1990). Nevertheless, given the lack of access to spawning habitat in the Tees, upstream
455 of High Force, the Balder and Lune tributaries had been among the most important spawning and
456 nursery areas for anadromous salmonids, but are now largely lost habitat due to impoundment.
457 Moreover, these dams have prevented the downstream transport of gravel for many decades,
458 causing the deterioration of salmonid spawning habitat downstream (Environment Agency, 2010;
459 Kondolf, 1997).

460

461 Following increased demand for construction in the early 20th Century, gravel and sand extraction
462 across the UK rapidly increased (Archer, 2003b). Gravel extraction from the Tyne and its tributaries
463 covered nearly 20 km river distance, and about 4.6 million tonnes of gravel were removed from the
464 river channel between 1891 and 1972 (Archer, 2003b). Extensive gravel extraction also occurred in
465 the middle Wear between 1945 and 1968 (Wishart et al., 2008). Gravel extraction along the Tyne
466 and Wear channels ceased almost entirely in the early 1970s but has had several likely negative
467 impacts, increasing upstream erosion and channel widening, and reducing gravel habitat for
468 spawning by fishes such as salmon, trout, lampreys, and minnow (Brown et al., 1998; Kondolf,
469 1997). Gravel extraction altered the middle Wear's river channel from braided to single thread
470 planform (Wishart et al., 2008), and has led to reduced summer flow and an increase in winter
471 spates over recent decades, potentially affecting fish migration and salmon survival following
472 hatching (Environment Agency, 2008).

473

474 **3.3 Recovery of water quality and habitats in the study rivers**

475 The first major period of water quality improvement in northeast England occurred during the 1960s-
476 1970s. During this period there was a large-scale reduction in heavy industry in England as it
477 relocated overseas, resulting in pollution abatement in former industrial areas, combined with wide-
478 scale investment in urban wastewater treatment of sewage and tighter regulation of industrial and
479 commercial waste releases (Langford et al., 2012; Mawle & Milner, 2003; Ord, 1988). A second
480 period of water quality improvement followed the privatization of the English water and sewerage
481 industry in 1989, combined with the formation of an independent environmental regulator, the
482 National Rivers Authority, and subsequently the Environment Agency, since when there has been
483 stronger regulation and enforcement of water quality (Mawle & Milner, 2003; Whelan et al., 2022),
484 although the adequacy of this has come under public scrutiny in recent years. Some would argue
485 that privatisation of the water industry enabled greater investment in sewage treatment and disposal
486 than would have been likely under public ownership (Meredith, 1992; Whelan et al., 2022); this may
487 have been the case for improvements to sewage treatment in urban areas of the study rivers under
488 the auspices of Northumbrian Water plc.

489

490 Lead mines in the North Pennines declined in the early 20th Century and closed by the mid-20th
491 Century (Kincey et al., 2022). As a result, water quality in the upper reaches of the rivers improved
492 markedly in the late 20th Century (Hynes, 1960; McParlin, 2011) though metal contamination and
493 ecological impacts remain high in places (Armitage et al., 2007). During the 20th Century the coal-
494 mining industry in northeast England moved eastwards as coal seams were exhausted. Two thirds
495 of the area's coal mines that were operational in 1960 had closed by 1970 and the last colliery in the
496 Durham coalfield closed in 1994 (Benyon & Hudson, 2021). Because of the importance of
497 groundwater aquifers and abstraction from the River Wear for public water supply, minewater
498 pumping has continued in northeast England so as to prevent rising levels in mine shafts from
499 contaminating groundwater and surface water (Johnston et al., 2008). This has, to date, avoided the
500 worst environmental impacts of coal mine flooding in the region. There has also been increased
501 treatment of pumped mine water, and of mine water waste, including leachate from spoil heaps,
502 amidst debate about the potential environmental impacts of unmanaged coal mine water release
503 (Johnston et al., 2008). Many of the spoil heaps from coal processing and coke works that closed in
504 the 20th Century underwent reclamation, mostly by partial levelling and tree and grassland planting,
505 usually combined with public recreational access. Spoil heaps from earlier mining have mostly been
506 allowed to self-heal except where obvious pollution has been evident.

507

508 **3.3.1 Recovery of the Tyne**

509 Coke works and related wastes such as tar acids and phenols in the Tyneside area declined in the
510 mid-20th Century and the last closed in the 1980s, with sites cleared and decontaminated, a pattern
511 repeated for Tyneside steelworks, shipbuilding, leadworks and chemical works. Sewage discharge
512 remained a problem and required the building of large sewage capture, transport and treatment
513 systems. On Tyneside, a major sewage treatment work scheme ('Howdon') was proposed in 1964,
514 comprising a sewer network to be constructed along both banks of the Tyne to capture sewage from
515 more than 200 outfalls, with wastewater transported to a new sewage treatment plant (Foster,
516 2003). Visual pollution, oxygen-demanding waste and smell in the Tyne estuary reduced greatly
517 after the work was completed in the early 1980s, but waste water, following secondary (activated

518 sludge) treatment, is still returned to the estuary, rather than to the open sea (Foster, 2003).
519 Historically this has caused oxygen-demanding waste (albeit at a much lower level) to be retained
520 within the estuary, due to tidal action, especially during periods of low river flow, particularly during
521 summer.

522

523 As estuarine pollution reduced in the study rivers, rod catches of salmon and sea trout increased in
524 the 1980s (detailed in section 3.4). From 1982 to 1988, increased numbers of fish species were
525 recorded during beam trawling in the Tyne estuary (Pomfret et al., 1988) and increased flounder
526 catches were recorded in the upper Tyne estuary (Hugman et al., 1984; Pomfret et al., 1988).
527 However, during summer, low dissolved oxygen levels in the estuary caused fish kills and the
528 problem persisted through the 1990s. In 1990, water quality in 30% of the Tyne estuary was
529 classified as 'poor' (Mawle & Milner, 2003). Salmon that remained in the cool estuary, particularly
530 during summer low flows with warm river water, often became stressed and susceptible to
531 pathogenic infections (National Rivers Authority, 1994, 1995a). In summer 1994, about 500 adult
532 salmonids died in the Tyne estuary (National Rivers Authority, 1994) due to low dissolved oxygen
533 concentrations and related stressors, a pattern repeated the following summer (Environment
534 Agency, 1996; National Rivers Authority, 1995a, 1995b). Reduced fish abundance and species
535 richness were recorded in the lower Tyne estuary during beam trawling surveys between 1982 and
536 1996, reflecting a decline in water quality there, due to increased discharges from the Howdon
537 Sewage Treatment Works (Gill et al., 2001).

538

539 Early efforts were made to improve river connectivity across the study rivers prior to 1900, including
540 the building of several salmon ladders at weirs, facilitated by acts of parliament supporting salmon
541 and freshwater fisheries. In 1862, Bywell weir, on the Tyne, was removed due to the complexity of
542 building a fishway there. In 1870, Woodburn weir was removed, and sea trout were observed
543 upstream in the River Rede for the first time since its construction (Marshall, 1992). Overall, there
544 was an increase in the number of anthropogenic instream barriers installed in the study rivers and
545 their tributaries throughout the late 19th and 20th centuries. Since the 1960s when salmon recovery
546 in these rivers began, there have been efforts to improve fish passage, mainly for salmonids and

547 mostly through the provision of 'technical' upstream fishways (i.e. heavily engineered upstream fish
548 passes). A summary of the fishways, passage easements and barrier removals on the study rivers is
549 given in Table S2.

550

551 Chemical water quality has improved across much of the Tyne catchment since the 1970s (Table
552 S3; Sun, 2021), both for organic pollution indicators and for heavy metals. Efforts to categorise
553 water quality of English rivers comprised the General Quality Assessment of the NRA/EA and
554 latterly through the EU Water Framework Directive (WFD) and its successor since Brexit. In 2015
555 and 2016, most Tyne WFD water bodies were classified as 'good' for chemical status (Table 1).
556 Since 2016, new substances (e.g. perfluorooctane sulfonate) have been added to the assessment
557 list and stricter standards have been developed for contaminants, which reflect the extent of these
558 chemicals in the environment more accurately (Environment Agency, 2020c). The introduction of
559 these meant that all water bodies in the Tyne catchment failed to achieve good chemical status in
560 2019 (Environment Agency, 2020c). The ecological condition in the Tyne improved slightly between
561 2015 and 2019, as some water bodies moved from poor to moderate status. However, 66.7% of
562 Tyne water bodies [82/123] failed to reach good ecological condition in 2019, with the greatest
563 pressures coming from hydromorphological modification and pollution from abandoned mines (Table
564 S4). In addition, pollution from rural and urban areas contributed to pressures on water bodies.

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576 Table 1. Ecological and chemical classification for surface waters in the Tyne, Wear and Tees
 577 catchments in 2015, 2016 and 2019 (cycle 2 of Water Framework Directive). Since 2016, new
 578 substances (e.g. perfluorooctane sulfonate) have been added to the assessment list and stricter
 579 standards have been developed for contaminants, which reflect the extent of these chemicals in the
 580 environment more accurately (Environment Agency, 2020c), resulting in none of the water bodies
 581 being graded 'good' for their chemical status.

Catchment			Ecological status or potential					Chemical status	
Basin	Year	Number of water bodies	Bad	Poor	Moderate	Good	High	Fail	Good
Tyne	2015	123	0	26	59	38	0	10	113
	2016	123	0	24	63	36	0	14	109
	2019	123	1	14	67	41	0	123	0
Wear	2015	64	2	10	44	8	0	11	53
	2016	64	0	14	44	6	0	8	56
	2019	64	1	15	42	6	0	64	0
Tees	2015	87	11	17	49	10	0	6	81
	2016	87	7	19	50	10	1	7	80
	2019	87	9	18	47	13	0	87	0

582

583 3.3.2 Recovery of the Wear

584 Since 1980, following the closure of coal and metal mines, and associated activities, particularly
 585 coke making, water quality in the Wear improved (Milner et al., 2004); these patterns are illustrated
 586 across the Wear catchment since the 1970s (Table S5; Sun, 2021), both for organic pollution
 587 indicators and for heavy metals. Overall improvements to sewage treatment have also led to
 588 improvements in water quality of the Wear (Environment Agency, 2008; Milner et al., 2004). In
 589 recent decades efforts have been made to reduce diffuse pollution (e.g. from agriculture, urban
 590 areas and roads) (Environment Agency, 2008) and remove nutrients (especially phosphorus) from
 591 STW effluent (Environment Agency, 2008). In the face of public protests, increasing efforts have
 592 been made across the Wear, as well as the Tyne and Tees, catchments to reduce problems of
 593 uncontrolled sewage release from Combined Storm Overflows, intended to occur only during

594 intense rainfall (affording high dilution), but found to occur frequently during normal and dry periods
595 also, breaching regulatory standards.

596

597 Similar to the Tyne, the majority of sites for chemical status were 'good' in the Wear in 2015 and
598 2016 (Table 1), but after new substances were added to the assessment list and stricter standards
599 applied, all Wear water bodies failed to achieve good chemical status in 2019 (Table 1). No clear
600 changes in the ecological status were found from 2015 to 2019, 58/64 (90.6%) of water bodies still
601 failed to reach good ecological condition in 2019, with the greatest pressures coming from
602 wastewater, urban and minewater pollution sources and from hydromorphological modification
603 (Table S6). There is still a long way to go in the recovery of the Wear, even if dramatic
604 improvements have been made in recent decades.

605

606 **3.3.3 Recovery of the Tees**

607 Since the 1970s, metal industries along the Tees estuary have declined (Smurthwaite, 2006)
608 including loss of most of the steel industry, with the largest steelworks closing in 2015 (Evenhuis,
609 2018). The chemicals industry on Teesside has slowly declined in recent decades (Evenhuis, 2018)
610 and industrial discharges have been controlled increasingly rigorously over the last half century
611 (Tapp et al., 1993; Warwick et al., 2002). To control domestic sewage pollution, large interceptor
612 sewers were built on both sides of the river for primary treatment of sewage at Portrack STW,
613 constructed in 1985, on the left bank of the estuary (Nelson, 2003). Until then, untreated sewage of
614 over 400,000 people was discharged to the estuary (Shillabeer & Tapp, 1989). There was a
615 dramatic decrease in Tees estuary BOD and ammonia during the 1970s, with these levels
616 maintained in the 1980s and 1990s (Tapp et al., 1993; Warwick et al., 2002). By 1996, it was
617 estimated that pollution levels in the estuary had reduced by 90% compared with 1970 (Watson &
618 Davis, 1995). Decreases in heavy metals were recorded in the Tees estuary and its surface
619 sediments since the 1970s (Davies et al., 1991; Jones & Turki, 1997). Following water quality
620 improvement, the Tees estuary's benthic fauna became more diverse between 1979 and 1990
621 (Tapp et al., 1993). Apart from chronic pollution, frequent acute pollution incidents in the 1980s to
622 early 2000s in Tees tributaries such as the Skerne inhibited their ecological recovery. For example,

623 in February 2000, an unidentified substance entered the Skerne and led to a large fish kill (Jenkins,
624 2000) of more than 1180 dace, 800 chub, 35 trout and small numbers of roach, pike (*Esox lucius*),
625 perch (*Perca fluviatilis*), gudgeon, and grayling.

626

627 Similar to the Tyne and Wear, improvements in water quality occurred in the Tees over the period
628 1973 to 2020 (Table S7; Sun, 2021), with decreases in BOD and ammonia, characteristic of organic
629 pollution, and decreases in heavy metals. Most Tees sites were graded 'good' for chemical status in
630 2015 and 2016 in the EA's second WFD cycle (Table 1), but after new standards applied by the EA
631 (Environment Agency, 2020c), all Tees water bodies failed to achieve good chemical status in 2019
632 (Table S8). No clear changes in ecological status of the Tees occurred between 2015 and 2019 and
633 85% of water bodies failed to reach good ecological condition in 2019, with the greatest pressures
634 coming from hydromorphological modification, pollution from wastewater, rural and urban pollution
635 sources (Table S8).

636

637 Several river restoration projects have taken place in the Tees catchment since the 1990s. The
638 River Skerne Restoration Project aimed to demonstrate rehabilitation of channel hydromorphology
639 by reinstating meanders and riffles to a 2-km section (Holmes & Nielsen, 1998; Vivash et al., 1998).
640 One year after restoration, plant species richness within the restored reach increased by 43%
641 (including nine intentionally reintroduced plants) compared to reference conditions (River
642 Restoration Centre, 1999). However, two years after restoration, the reach remained dominated by
643 pollution-tolerant invertebrates including water hoglouse (*Asellus aquaticus*), leeches and water
644 snail species (River Restoration Centre, 1999). Fish species richness increased slightly in the
645 restored reach two years after restoration, possibly due to the newly installed fishway immediately
646 downstream in autumn 1996.

647

648 **3.4 Recovery of salmon and sea trout in the Tyne, Wear and Tees**

649 Data on adult salmon and sea trout catches and counts provide an indicator of post-industrial river
650 recovery for northeast England's rivers; information for the Tyne is more complete than for the Wear
651 and Tees. Historically, commercial nets were fished in the rivers and along the coast, while rod

652 fisheries occurred in the rivers. In 1934, the net fishery in the Tyne (seine nets and some T nets
653 [employing a leader and net enclosure]) was banned in order to help restore the rod fishery
654 (McParlin, 2011). The northern coast net fishery (stretching southwards from the River Tweed, one
655 of the most productive UK salmon rivers, to the Tyne) fishery started to recover in the 1950s (Figure
656 S3) and salmon capture per licence reached pre-1870s level by the end of the 1960s (Champion,
657 2003). On the southern coast (Tyne to Tees) sea trout catches started to recover in the early 1960s
658 and salmon started to recover in the late 1960s (Figure S3).

659
660 Because adult salmon and sea trout tend to migrate southwards along the northeast English coast
661 before entering their natal river (Potter & Swain, 1982), the associated coastal net fishery, especially
662 the use of drift nets, has been a 'mixed stock' fishery, exploiting multiple natal river stocks (*sensu*
663 Begg et al., 1999). This made effective management and conservation of salmon and, to a lesser
664 degree, sea trout stocks, contentious. In order to reduce exploitation of mixed stocks, the EA (and
665 its predecessor the NRA) used 10-year Net Limitation Orders in 1992 and 2002 to limit the number
666 of licences. A permanent buyout of most drift net licences in 2003 (Environment Agency, 2008) was
667 followed by closure of the English drift net fishery in 2019 (Environment Agency, 2020b). At the
668 same time, the beach net fishery, including T nets, was closed for salmon as a conservation
669 measure, but northeast English fishermen were allowed to continue to fish for sea trout and
670 allocated larger quotas as compensation (Environment Agency, 2020b). Experiments are ongoing to
671 minimize salmon capture in modified T-nets (Environment Agency, 2020c).

672
673 As water quality improved in the 1960s and 1970s, the Tyne salmon population started to recover
674 and the annual rod catch dramatically increased from the late 1980s (Figure 3). The rod catch of
675 salmon increased by three orders of magnitude, from three in 1960 to 5,663 in 2011, while that of
676 sea trout increased from 15 in 1960 to 2,687 in 2010 (Figure 3). Since 2010 annual Tyne rod
677 catches of salmon and sea trout, though variable, have shown a decreasing trend, despite reducing
678 commercial licenses and quotas, and eliminating these for salmon in 2019. The rod catch per unit
679 effort (CPUE, catch per licence day) of salmon in the Tyne increased between 1993 and 2004, then
680 fluctuated between 2004 and 2022, with a peak in 2011 (Figure 4). Sea trout CPUE increased

681 between 1993 and 1998, then fluctuated between 1999 and 2022.

682

683 Salmon rod catches increased in the Wear from the 1970s to the 2010s (Figure 3), as water quality

684 improved, a pattern similar to that for the Tyne. Sea trout catches in the Wear followed a similar

685 pattern, with very low rod catches of sea trout in the 1950s, increasing to several hundred per year

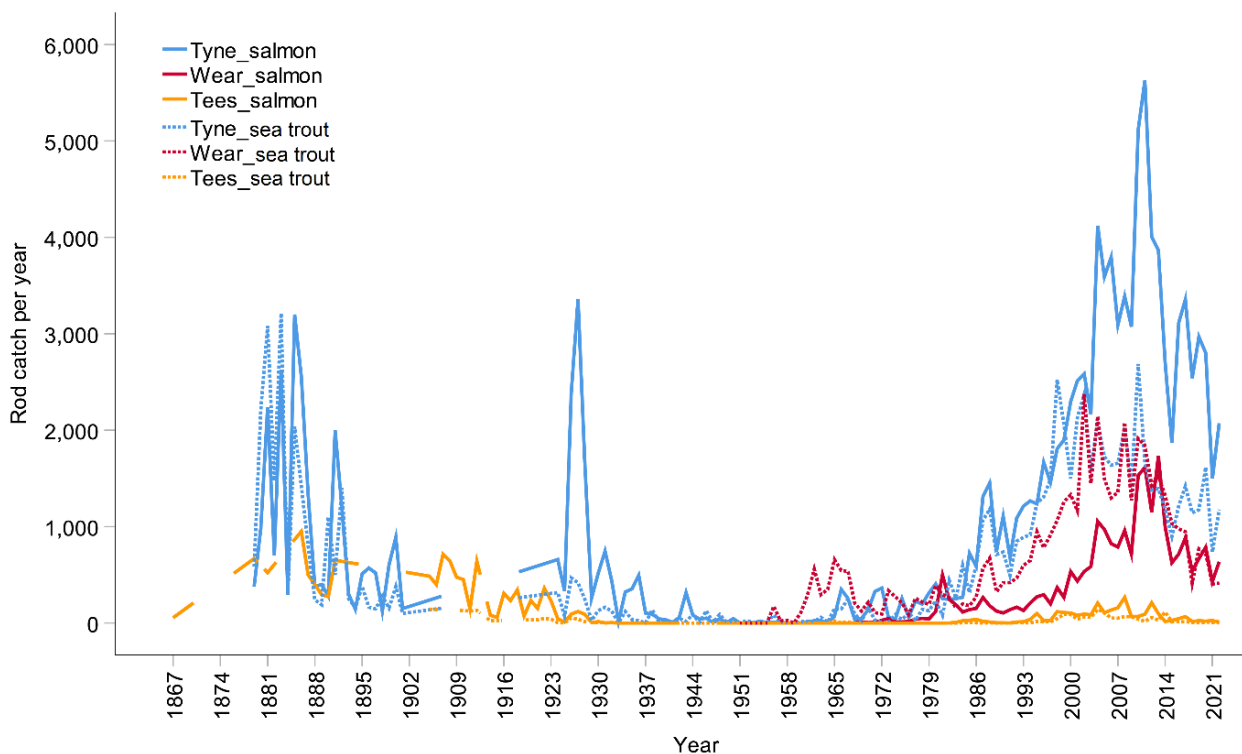
686 from 1960 to 1989. Catches of sea trout in the Wear peaked in 2002, when 2,374 were caught.

687 Since 2010, Wear salmon and sea trout rod catches have decreased, and catches in 2017 were

688 nearly half those of 2010. Rod CPUE of salmon in the Wear increased between 1993 and 2008 and

689 has since fluctuated, while sea trout CPUE increased between 1993 and 1998, fluctuated between

690 1999 and 2009 and has declined since then (Figure 4).



691

692 Figure 3. Annual declared rod catch of salmon and sea trout in the Tyne, Wear and Tees from 1867

693 to 2022. Note that before 1950 rod catch data suffer from gaps in historic recording (no data for

694 River Wear) and can serve only as a crude indication of temporal patterns of catch.

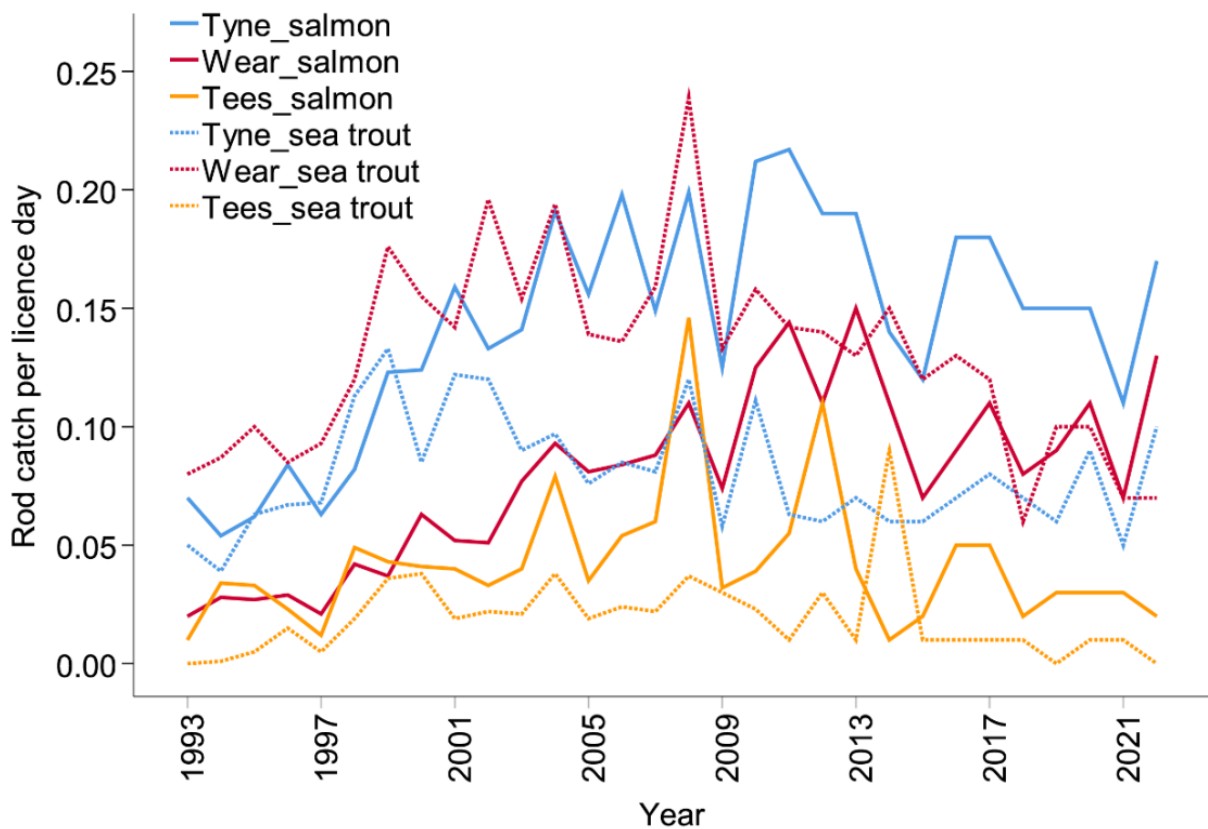
695

696 From the 1930s until the 1980s, when the Tees began to recover, there was no salmon rod fishery

697 there due to severe pollution and resultant lack of anadromous salmonids. Reduction in estuary

698 pollution increased oxygen levels, making the passage of salmonids possible (Environment Agency,

699 1999). Annual rod catches in the Tees increased slowly in the 1980s and 1990s, and peaked in
700 2008, when 267 salmon were caught. The declared sea trout rod catch followed a similar pattern,
701 peaking in 2004 with a catch of 143 fish. Tees salmon CPUE increased between 1993 and 2008,
702 fluctuated between 2009 and 2013 but has declined since. The Tees has much lower CPUE for
703 salmon and sea trout than for the Tyne and Wear (Figure 4), suggestive of lower abundance in the
704 Tees.
705



706
707 Figure 4. Rod catch per licence day fished of salmon and sea trout in the Tyne, Wear and Tees from
708 1993 to 2022.

709
710 A hatchery was built at Kielder, as a legal requirement, to stock the Tyne with salmon in
711 compensation for the loss of wild salmon production above Kielder dam (Cave, 1985) and has
712 operated since 1979. Broodstock collected from below the dam are stripped of eggs and milt, and
713 released back to the river. The legal mitigation is to stock a total of 160,000 0+/1+ juvenile salmon in
714 the Tyne annually (Milner et al., 2004). Between 1989 and 2017, an average of 379,898 (range,

715 46,000 - 638,898), comprising fry, parr (age 0+ and 1+) and smolts were stocked in the Tyne (Milner
716 et al., 2004). Estimates of the long term (1980-2000) contributions of stocked Kielder salmon,
717 measured from returns of coded wire micro-tagged parr, to the northeast commercial fishery and the
718 Tyne rod catch were 1.5% (range 1.2-2.0%) and 6% (range 3-14%) respectively (Milner et al.,
719 2004). An estimated overall hatchery contribution (1980–1986) of up to 20% of the cumulative adult
720 run is suggested to have aided initial recovery of Tyne salmon (Milner et al., 2004). From 1989 to
721 2017, a total of 10.9 million salmon were released in the Tyne. Such stocking may no longer be
722 necessary from an ecological perspective, and may be counterproductive (Lennox et al., 2021),
723 although the Kielder hatchery's activities increasingly focus on wider conservation benefits such as
724 supporting pearl mussel (*Margaritifera margaritifera*) restoration, by rearing and harvesting pearl
725 mussel glochidia on juvenile salmon hosts (Bolland et al., 2010).

726

727 Some excess salmon from Kielder hatchery were stocked in the Wear, but the small numbers
728 stocked suggest that the Wear salmon recovery was based on natural recolonization, probably
729 facilitated by strays from other rivers, especially the nearby Tyne and Tweed. Between 1985 and
730 1996, 1.29 million excess Kielder salmon were stocked in the Tees, mostly at ages 0+ (72%) and 1+
731 (22%) (Environment Agency, 2009). Among 102,794 tagged 1+ salmon released to the Tees from
732 Kielder by 1994 (Jowitt & Russell, 1994), six were captured by Tees anglers and 411 were reported
733 by other rod and net fisheries by 1995 (Shelley, 1994). Unlike the Tyne, salmon stocking appears to
734 have had insignificant benefits on Tees salmon recovery, although the paucity of data limits
735 confidence in this. There has been recent interest from landowners for a Tees salmon hatchery, but
736 this has been resisted by the EA, based upon contemporary evidence of the ecological value of
737 salmon stocking (Hansen, 2002; Lennox et al., 2021; Miller et al., 2004).

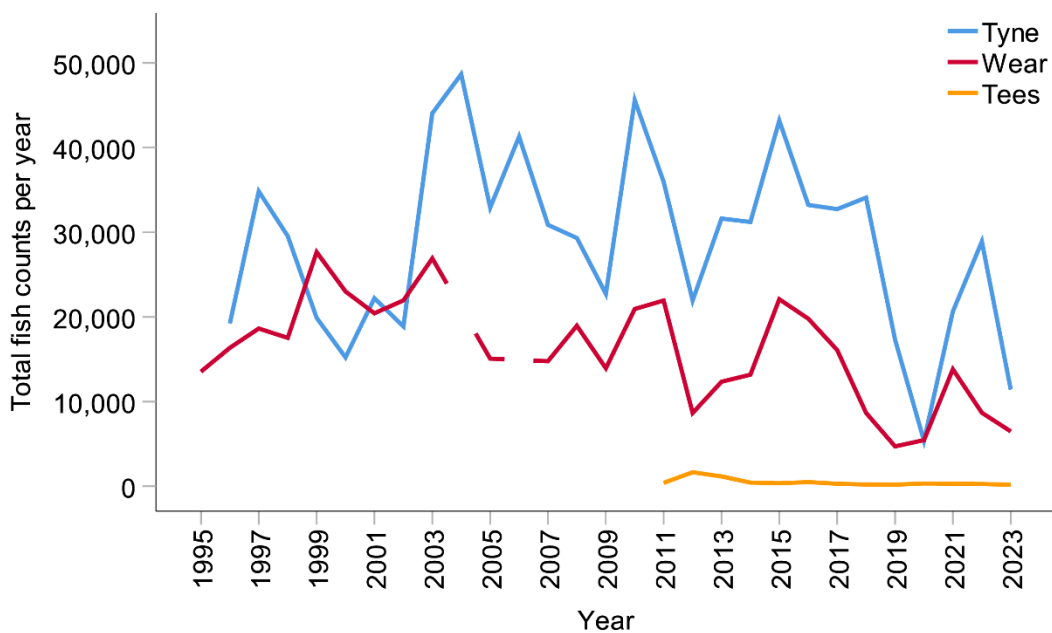
738

739 Since operation of the resistivity fish counter on the Tyne began in 1996, annual upstream fish
740 counts (salmon and large trout >40 cm long, primarily sea trout) have ranged from 15,219 (2000) to
741 48,668 (2004) (Figure 5), but only since 2003 was video capture added to the counter to help
742 distinguish species. Not all salmon and sea trout are recorded by the Tyne counter, as fish can
743 ascend the weir directly at high flows; the EA assumes 10% bypass the counter (P. Rippon pers.

744 comm). Fish counter records on the Wear commenced in 1994 and a second counter at the
 745 upstream exit of a new fishway on the same set of weirs started operating in February 2015. Total
 746 fish counts fluctuated between 1994 and 2023 (Figure 5), with the highest recorded annual total of
 747 27,658 fish occurring in 1999, and a decreasing trend in recent years, reflecting the pattern of
 748 decreased salmon and sea trout rod catches and sea trout CPUE.

749
 750 The recovery of anadromous salmonids in the Tees started later than the Wear and Tyne, becoming
 751 evident in the 1980s until the mid-2000s (Environment Agency, 2009), since when improvement has
 752 halted, or even reversed (Figures 3-5). This is reflected by electronic fish counts (Figure 5), one to
 753 two orders of magnitude lower in the Tees than the Tyne and Wear, even though these are partial
 754 counts of each river's population. Since peaks in Tees salmon / sea trout rod catches in 2008 and
 755 2012, the latter associated with an upstream fishway count of 1,661, Tees annual fish counts have
 756 numbered only several hundred, although fish also pass by other routes.

757



758
 759 Figure 5. Annual upstream counts of migratory adult salmonids (salmon plus sea trout) from 1995 to
 760 2023 on the Tyne (Riding Mill), Wear (from February 2015 onwards these represent a combined
 761 count from both Framwellgate and Freemans counters, Durham) and Tees (Tees Barrage). Note:
 762 the Wear fish counter did not operate during the salmonid upstream migration periods in 2004 and
 763 2006. The Tyne fish count data from April to July 2020 were unavailable so 2020 data are a

764 minimum.

765

766 Several acoustic tracking studies have measured passage success of adult anadromous salmonids
767 at the Tees Barrage. In 2002, out of 103 adult salmon and 53 sea trout caught from the navigation
768 lock, tagged and released downstream, three salmon and two sea trout passed the barrage, one
769 through the canoe slalom and four by the barrage gates (Environment Agency, 2009). Between
770 2008 and 2013, 199 salmon and 38 sea trout, captured in T-nets at the estuary mouth, were tagged
771 and tracked. Eleven out of 80 fish (13.8%) that approached the barrage passed it, 10 by the barrage
772 gates and one by a fishway (Moore & Potter, 2014). Over 30% of tagged fish are known to have
773 died and at least 14.7% were predated by seals (Moore & Potter, 2014). In recent years resident
774 seals, especially grey seal (*Halichoerus grypus*), have become an increasing cause of adult
775 salmonid mortality at the barrage, highly conspicuous to onlookers as those seals eat salmon close
776 to the barrage. Long fish-passage delays and repeated passage attempts due to the barrage's
777 obstructing effect put migrating salmonids at high predation risk.

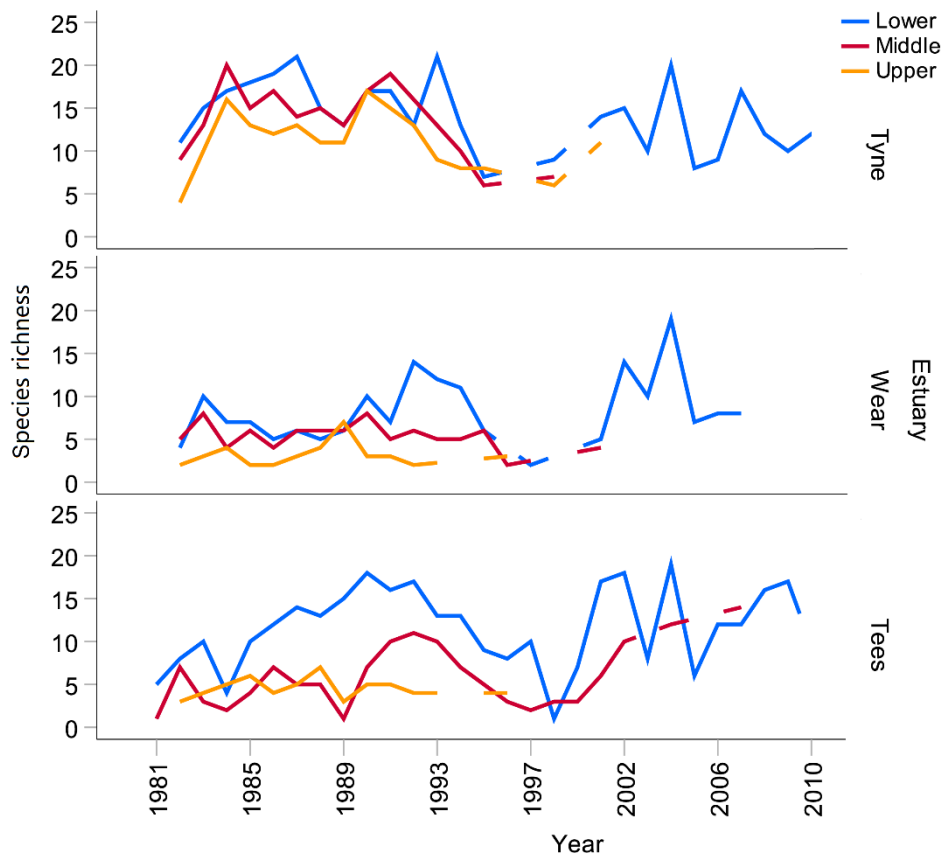
778

779 **3.5 Recovery of the Tyne, Wear and Tees fishes other than anadromous salmonids**

780 **3.5.1 Estuarine fishes**

781 Beam trawl data from the Tyne, Wear and Tees estuaries show wide variability in species richness
782 over time, possibly linked to cycles of change in the coastal environment. Close to maximal species
783 richness was already apparent in the Tyne by the late 1980s (Figure 6). The Wear estuary was
784 characterised by lower species richness than the Tyne or Tees. There is some evidence of an
785 increase in species richness between 1981 and 2010 in the Wear and also for the middle estuary
786 reach of the Tees (Figure 6). No records were made of smelt, formerly abundant in the Tees
787 estuary and lower river.

788



789

790 Figure 6. Historical changes in fish species richness in the estuaries of the Tyne, Wear, and Tees.

791 All data were obtained from standardised beam trawl surveys and the values represent the annual
 792 cumulative species richness from all samples taken.

793

794 Temporal changes in fish community structure of the three estuaries occurred over the period 1981

795 to 2010 (Figure 7). For the Tyne estuary, key species (85% cumulative contribution threshold)

796 significantly contributing to fish community changes, determined by Similarity Percentage Analysis

797 (SIMPER), between the 1980s and 2000s, were the clupeids, herring (*Clupea harengus*) and sprat

798 (*Sprattus sprattus*), both of which decreased in relative abundance (SIMPER, Table S9). In the

799 Wear estuary, over the same time period, herring and eelpout (*Zoarces viviparus*) decreased in

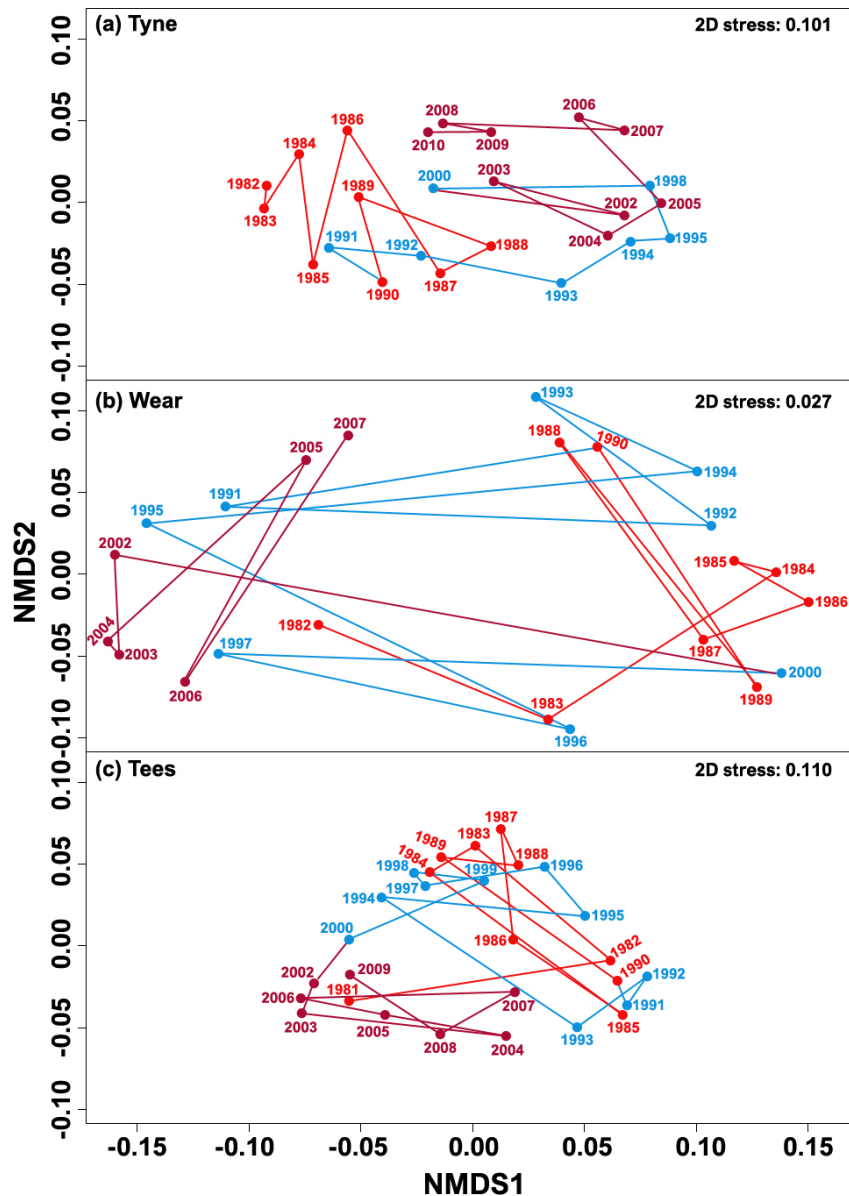
800 relative contribution, while pogge (*Agonus cataphractus*) and dab (*Limanda limanda*) increased

801 (SIMPER, Table S10). In the Tees estuary, significant contributions to fish community change

802 between the 1980s and 2000s comprised increased abundance of two flatfish species

803 (*Pleuronectes platessa*, *Limanda limanda*), pogge and cod (*Gadus morhua*) (SIMPER, Table S10).

804



805

806 Figure 7. Non-metric multidimensional scaling (NMDS) ordination plots of fish community structure

807 for estuaries of a) Tyne, b) Wear, c) Tees over the period 1981-2010. Generated using data

808 combined from lower (strongly brackish) and middle estuary (moderately brackish) samples, with

809 decadal data colour-coded for clarity

810

811 3.5.2 Freshwater fishes

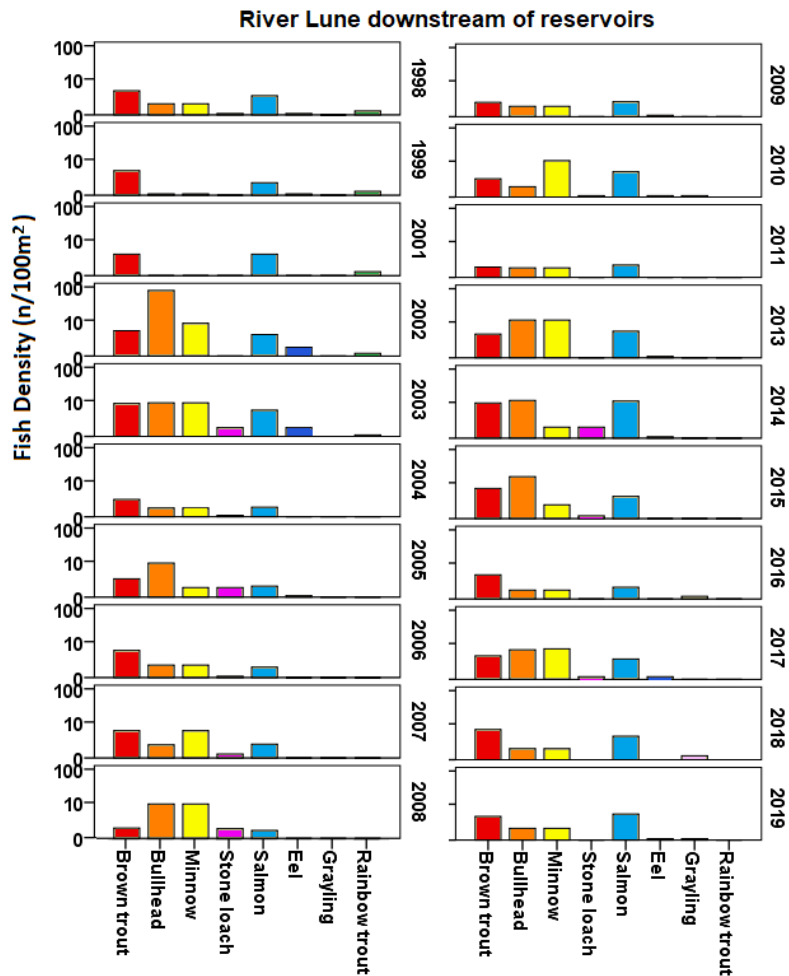
812 Species richness of freshwater fish decreases from the Tees, to Wear, to Tyne (Table S1),

813 associated with more cyprinid species in the south of the region. The extent to which this is due to a

814 latitudinal post-glacial dispersal constraint is unclear as fish species have been translocated for

815 food, recreation and fishing bait since at least Medieval times (Wheeler, 1969; Wheeler & Maitland,

816 1973). Several species, mostly cyprinids, but also perch and grayling, have been stocked into these
817 rivers in recent decades by the EA to support demand for recreational fisheries and to aid the
818 restoration of fish stocks from pollution (e.g. River Skerne, a lower Tees tributary). These actions
819 include stocking of barbel (*Barbus barbus*) to the Wear, for which there is no evidence that they
820 have ever been native in this river (Britton & Pegg, 2011; McParlin, 2011; Wheeler, 1969). Non-
821 native rainbow trout (*Oncorhynchus mykiss*) and native brown trout are also stocked into many
822 stillwaters within the study area, and the latter into some running waters for angling. Large numbers
823 of non-diadromous brown trout were stocked into all three rivers in the 19th and 20th centuries
824 (Owen et al., 1993; McParlin, 2011) but today they must be sterile triploid stock, unless obtained
825 from local broodstock approved by the EA. In the 1970s-2000s there was a small, self-sustaining
826 population of rainbow trout in the River Lune tributary of the Tees, that originated from Grassholme
827 Reservoir reservoirs, but these have now died out (Figure 8). Numerous ponds and lakes connected
828 by watercourses to the rivers also lose non-native species such as carp (*Cyprinus carpio*) during
829 floods but, currently, the rivers are too cool and fast-flowing for their successful reproduction. The
830 freshwater fish communities are therefore, a mixture of natural populations, ones supported by
831 stocking and escaped or deliberately translocated non-native species. Although there is little direct
832 evidence that non-native fishes in these rivers have impacted native fishes, good evidence exists
833 that non-native species such as signal crayfish (*Pacifastacus leniusculus*) are impacting salmonid
834 and bullhead recruitment within the Tees, as well as altering food webs (Galib et al., 2021, 2022).
835
836



837

838 Figure 8. Long-term variation of estimated minimum fish density (1995, 1999, 2007- 2019: single
 839 pass electric fishing; 1998: first run data from two pass electric fishing; 2001- 2006: first run data
 840 from three pass electric fishing; estimated three pass capture efficiency: trout 88.6%, salmon
 841 96.8%) at River Lune sites between 1995 and 2019 (log scale).

842

843 The study rivers all exhibit longitudinal zonation of freshwater fishes (Huet, 1959). In the upper
 844 reaches, fish communities are dominated by juvenile salmon and brown trout, minnow and bullhead
 845 (Figure 8; Figure S4). In the Tees and Wear, grayling and rheophilic cyprinids such as chub and
 846 dace are common in the middle and lower reaches (Figure S5; Owen et al., 1993). In the Tyne
 847 catchment, grayling (introduced) are present only in the Derwent and rheophilic cyprinids occur only
 848 in the lower Tyne. All three native species of lamprey are present in all three study rivers, with
 849 anadromous sea lamprey (*Petromyzon marinus*) and river lamprey (*Lampetra fluviatilis*) restricted
 850 mostly to the lower reaches, at least partly due to passage restrictions at weirs on the main rivers

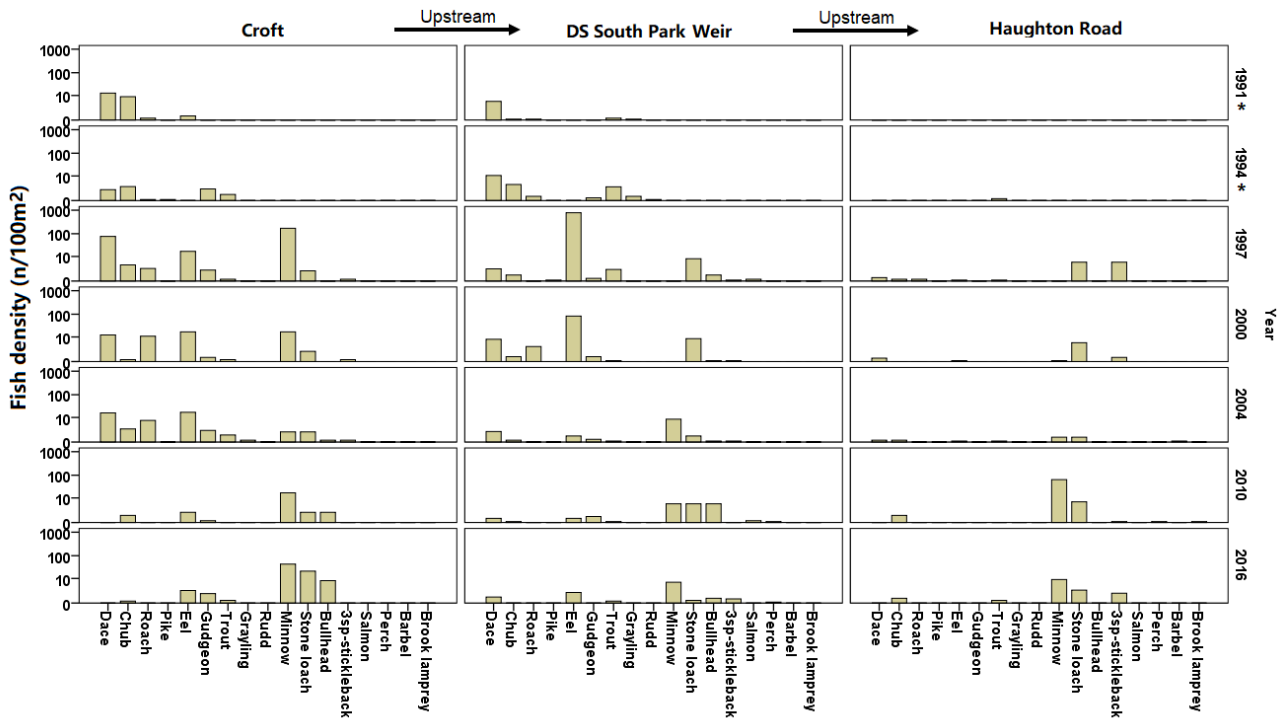
851 (Lucas et al., 2021), and brook lamprey (*Lampetra planeri*) occurring in many of the tributaries in the
852 middle and upper reaches (Table 1, Bracken et al., 2015). Upstream passage past the Tees
853 barrage is problematic for river and sea lamprey (M. Lucas, unpublished data). After the
854 construction of the Tees Barrage, roach, bream (*Abramis brama*) and chub fry numbers increased in
855 the impounded section (Welton et al., 1999). Older dace, chub and roach densities also increased in
856 the impounded reach post-barrage (Welton et al., 1999), reflected in persistent increases in anglers'
857 catches.

858

859 Pollution and hydromorphological modification were most intense in some downstream tributaries,
860 which were effectively fishless throughout much of the 20th Century, but native fish communities
861 persisted in the middle and upper river reaches. A survey of the Tees catchment in 1992 showed
862 that Clow Beck, the River Greta and Langley Beck, all in the middle Tees, held abundant brown trout
863 and other small species such as minnow, whereas no fish were caught in Billingham Beck and
864 Lustrum Beck in the lower catchment (Owen et al., 1993). While major efforts have been made over
865 the last 60 years to improve water quality in the freshwater reaches of the study rivers, it is only in
866 the last 30 years that major efforts have been made to restore hydromorphic diversity and natural
867 habitat, and to remove or mitigate fish passage barriers there (Holmes & Nielsen, 1998; Sun et al.,
868 2021; Sun, Tummers, et al., 2022; Tummers et al., 2016; Vivash et al., 1998). For fish communities,
869 increasing emphasis has been placed upon achieving reach-scale connectivity to facilitate natural
870 dispersal and recolonisation of weaker-swimming and smaller species, increasingly employing
871 barrier removal and nature-like passes (Sun et al., 2021; Sun, Tummers, et al., 2022; Tummers et
872 al., 2016). Many, though not all, of the tributaries have sufficient gradient and discharge variation to
873 institute natural habitat recovery (e.g. riffle formation), but achieving natural habitat connectivity has
874 been afforded insufficient effort. To achieve ecological recovery at the landscape scale requires
875 achievement of good water quality, habitat and connectivity, usually at high cost and takes years or
876 decades for fish communities to recover, as evidenced in the River Skerne of the Tees (Figure 9)
877 and the River Deerness (Sun, Tummers et al., 2022) and Cong Burn of the Wear (Figure S6).
878 Nevertheless, in some cases, inexpensive removal of a single barrier can dramatically improve
879 stream habitat, fish passage and recolonization within a year, as shown for removal of a 2.4-m high

880 barrier at the tidal limit on Claxton Beck in the lower Tees catchment (Sun et al., 2021).

881



882

883 Figure 9. Long-term variation of estimated minimum fish density at River Skerne sites between 1991
884 and 2016. Fish density was calculated by single pass electro-fishing method. Note: abundance is on
885 log scale. No fish were caught in 1991 at the Haughton Road site. Minnow, stone loach, bullhead
886 and stickleback may have been present but were not recorded or counted in 1991 and 1994
887 surveys for all sites

888

889 4. Discussion

890 Historical ecology has a valuable role to play in assessing population trends of fish and other
891 species over time and in their conservation and recovery (Kroes et al., 2025; McClenachan et al.,
892 2024; Pierce et al., 2013; Poulsen, 2010; Thurstan, 2022). This study provides historical evidence
893 regarding anthropogenic impacts on three rivers in northeast England (Tyne, Wear and Tees) dating
894 back to before the First Industrial Revolution and shows the timeline of decline and recovery of fish
895 across the three rivers. Resurrection of the Tyne and Wear from open sewers with negligible Atlantic
896 salmon runs, to the first- and second-best salmon rivers (by abundance) in England and Wales is a
897 restoration achievement to be appreciated, but not underestimated. Factors that led to the near

898 extirpation of salmon and sea trout in these rivers were identified as pollution, overexploitation,
899 widespread river barriers and habitat modification. Information on changes in the abundance and
900 distribution of other native fish species in the study rivers has been fragmentary until recent
901 decades, but available evidence suggests they have benefitted from the restoration measures also.
902 Nevertheless, understanding the historical context and effects of the factors causing the decline and
903 recovery of fishes in industrially impacted rivers is a precondition for their restoration (Palmer et al.,
904 2005; Sun, Tummers, et al., 2022; Wohl et al., 2015). Common themes and lessons learnt can be
905 applied for the fish and fisheries on other impacted river systems which have been affected by
906 historic human activities (Langford et al., 2012; Lennox et al., 2021) and which are necessary for the
907 Emergency Recovery Plan for global freshwater biodiversity (Tickner et al., 2020).

908

909 **4.1 Patterns of decline and recovery**

910 The causes of decline in anadromous salmonids and modification of fish communities in northeast
911 English rivers attributed here align broadly with others in Europe and North America (Langford et al.,
912 2012; Lenders, 2017; Limburg & Waldman, 2009; Waldman, 2006;). Our study emphasizes the
913 combined effects of pollution, overexploitation, river barriers and habitat degradation on river fish
914 decline and recovery more than past studies on UK rivers (cf Doughty & Gardiner, 2003; Ikediashi et
915 al., 2012; Langford, 2012; Mawle & Milner, 2003). Comparable trajectories of decline are widely
916 exemplified by rivers such as the Rhine (Europe) and Hudson (United States). These have involved
917 multiple causes of cumulative decline, over protracted time periods (decades or centuries) and large
918 spatial scales (whole river basins) resulting in complex, landscape-scale, multi-decadal restoration
919 actions to try and reverse impacts (Lenders, 2017; Kroes et al., 2025; Waldman & Quinn, 2022).
920 The Rhine has been heavily modified for centuries (Lelek & Köhler, 1990; Raat, 2001) and seven
921 native fish species were extirpated by 1990 (Lelek & Köhler, 1990). Following centuries of decline,
922 Atlantic salmon is functionally extinct in the Rhine, despite attempts to restore it (De Groot, 1992,
923 2002; Lenders, 2017). The main reasons for the decrease of native fish diversity in the Rhine are
924 river barriers inhibiting migration, loss of intact floodplain, loss of spawning and nursery grounds due
925 to gravel extraction, domestic and industrial waste water pollution, and overfishing (De Groot, 1992;
926 Raat, 2001), although major ecological restoration measures are underway (Bölscher et al., 2013)

927 and achieving recovery of some anadromous species (Kroes et al., 2025). The Hudson River
928 suffered similar pressures, especially from sewage and industrial pollution for over 300 years, that
929 caused the decline of diadromous fishes such as American shad (*Alosa sapidissima*) and the
930 sturgeons *Acipenser oxyrinchus oxyrinchus* and *Acipenser brevirostrum* through the late 19th
931 and 20th centuries (Limburg et al., 2006; Waldman, 2006; Baker et al., 2006). Since the 1960s,
932 Hudson River water quality steadily improved due to better water treatment and waste regulation
933 (Brosnan et al., 2006), and major physical habitat restoration measures are underway here and in
934 other industrially impacted US rivers (Miller, 2013).

935

936 Fish vanished in parts of many British rivers within industrial and urban areas during the second half
937 of the 19th Century and early 20th Century (Doughty & Gardiner, 2003; Mawle & Milner, 2003). The
938 partial recovery of fish stocks, including anadromous salmonids, to many of these is widely
939 attributed to pollution abatement, and to a lesser degree, reduced commercial exploitation and
940 improved access past river barriers (Doughty & Gardiner, 2003; Ikediashi et al., 2012; Mawle et al.,
941 1985; Mawle & Milner, 2003). Ecological recovery of the study rivers followed a similar pattern,
942 reflecting industrial decline in the 20th century, reduced industrial pollution and coinciding with
943 increased sewage treatment and more stringent environmental regulation of point source pollution.
944 However, the slow recovery of anadromous salmonids in the Tees is due, at least in part, to the
945 estuarine Tees Barrage, which is a major barrier and mortality and predation hotspot, a pattern
946 found widely at river barriers (Blackwell & Juanes, 1998; Boulêtreau et al., 2018; Palder et al.,
947 2026).

948

949 In some regards, the recovery of anadromous salmonids observed on the Tyne and Wear is
950 exceptional within the UK and industrialized Europe and North America. Major efforts to improve
951 water quality and connectivity within previously industrialized Atlantic rivers of the US have not
952 resulted in recovery of naturally spawning Atlantic salmon, even though other species such as
953 striped bass (*Morone saxatilis*) and river herrings have recovered well (Waldman & Quinn, 2022),
954 suggested to be due, in part, to climate change effects in rivers and at sea, impeding Atlantic
955 salmon recovery (Lennox et al., 2021). The River Trent's (central and eastern England) ecology has

956 recovered substantially from intense industrial and urban pollution (Langford et al., 2012), but wild
957 Atlantic salmon and sea trout populations in the Trent are in the early stages of recovery (Cefas et
958 al., 2024), lagging 3-4 decades behind the Tyne and Wear. Thermal pollution from numerous power
959 stations contributed to summer-time hypoxia in the lower Trent until the 1980s (Langford et al.,
960 2012), but the main problem for restoration of the Trent's anadromous salmonids has been the very
961 large number of river barriers along the migration route, mostly linked to its industrial heritage (Nunn
962 & Cowx, 2012). These barriers necessitate massive investment to provide fish passage solutions,
963 yet even efficient passage per barrier results in poor cumulative upstream migration success in
964 rivers with many barriers such as the Trent (Jubb et al., 2023; Silva et al., 2018). This situation is
965 even more evident for the River Thames, England, where Atlantic salmon was extirpated by the
966 1830s (Griffiths et al., 2011). Although the Thames fish fauna has recovered dramatically since the
967 1970s (Colclough et al., 2002), due principally to pollution reduction (Griffiths et al., 2011), high-
968 profile attempts to restore Atlantic salmon have failed due to the large numbers of barriers, diffuse
969 pollution, habitat degradation and climate warming (Griffiths et al., 2011). Some once common fish
970 species in the Thames such as European river lamprey (*Lampetra fluviatilis*) have failed to
971 recolonise, even though water quality is adequate for this species (Lucas et al., 2020).

972

973 In many countries, river impoundments have caused widespread impacts on river hydrology,
974 connectivity, fish migration and biological invasions (Chen et al., 2023; Johnson et al., 2008; Lynch
975 et al., 2024). Water transfers are also regarded as ecologically risky due to potential pathogen
976 spread and pollutant transfer (Zhuang, 2016). Yet in the studied rivers, impoundments in the upper
977 reaches have had some important benefits for sustaining native river fish communities in a context
978 of increasing water demands and the effects of climate change (Charlton & Arnell, 2011;
979 Environment Agency, 2025). Even during very dry summers, the capacity to transfer water from
980 Kielder to the Wear and Tees has meant that excessive potable water extraction from the rivers or
981 aquifers has been unnecessary, and provides a future-proofed climate resilience (Lynch et al., 2024)
982 beneficial to the ecological integrity of the rivers that many other catchments lack, even in the UK,
983 with its relatively high rainfall.

984

4.2 Adequacy of restoration in the Tyne, Wear and Tees

Sensitive management can achieve rapid recovery of depleted salmon populations (Koed et al., 2020; Lennox et al., 2021). In the Tyne and Wear, salmon and sea trout rod catches increased from the 1960s to ~2010, but have declined somewhat since, reflected also in upstream fish counts. These metrics suggest these rivers may have reached production capacity for the degree of hydromorphic, chemical and ecological recovery achieved. Atlantic salmon marine survival has declined in recent years (Gillson et al., 2022; Thorstad et al., 2021), and in 2023 the Great Britain subpopulation was listed as Endangered. As a conservation measure salmon commercial netting along the northeast English coast was banned in 2019, and since 1999 all English salmon caught on rod and line before 16 June ('spring' salmon) must be returned. To offset impacts on the commercial fishery the regulatory agency (EA) increased catch quotas for sea trout, a questionable decision given that sea trout abundance has decreased recently in the Tyne and Wear. Although water quality, habitat quality and overall availability of spawning and juvenile habitat are greater in the Tyne than the Wear, substantial room for improvement remains for both rivers given the high number of waterbodies failing to reach 'good' ecological quality (Table 1). The recent declines in adult salmon and sea trout abundance in these rivers also give cause for concern that restoration efforts are being impacted by wider stressors such as eutrophication and toxins in diffuse pollutants (Ramage et al., 2025; Whelan et al., 2022). Recovery of the Tees started about 30 years later than the Tyne and Wear catchments and it is still not achieving its current conservation limit for salmon egg deposition (Cefas et al., 2023; Environment Agency, 2009b). Since 2019 'catch and release' of all rod-caught salmon has been mandatory on the Tees. Inadequate passage at the Tees Barrage remains a major impediment to the recovery of anadromous salmonids in the Tees. Neither have smelt recolonised the Tees since their extirpation in the early 20th century. The Tees Barrage needs to accommodate delay-free bidirectional passage of all migratory species (Silva et al., 2018), whereas currently only upstream passage of salmonids is being emphasized. From a broader global perspective, a much more holistic and integrated approach is needed towards restoring river connectivity, including for future-proofing against climate change (Lynch et al., 2024; Stoffers et al., 2024).

1014 Although there has been good recovery of salmon and sea trout in the Tyne and Wear, challenges
1015 remain for fuller restoration. The EU WFD has been a strong driver of this in recent decades and
1016 since Brexit a parallel pathway has been followed. Only a small proportion of tributaries in these
1017 catchments have achieved good ecological status and all waterbodies failed the chemical status in
1018 2019, after new substances were added to the chemical quality assessment and new standards
1019 were introduced. The long-term impacts of many of these pollutants are as yet not fully understood
1020 (Alharbi et al., 2018; Johnson et al., 2013). In rural areas of the catchments, diffuse pollution from
1021 poor land management is causing siltation of watercourses and contributing to excessive
1022 phosphorus and ammonia levels (Environment Agency, 2017, 2020a). Instream barriers,
1023 channelisation, eutrophication and fine sediment remain major causes of not achieving good
1024 ecological status across the study catchments, despite some valuable improvements (Sun,
1025 Tummers et al., 2022).

1026

1027 Although the study rivers contain numerous non-native species, including fishes, these generally
1028 seem not to have caused major impacts on the fish fauna, or at least at a much lesser magnitude
1029 than the effects of pollution, habitat degradation, river barriers and overexploitation. Nevertheless,
1030 invasive crayfish have been played a part in the loss of native crayfish from the Tees and have been
1031 shown to have a demonstrable impact on brown trout and bullhead recruitment (Galib et al., 2021,
1032 Galib et al., 2022), and a precautionary approach is strongly advised for managing non-native
1033 species risks in all river systems (Britton et al., 2011; Lynch et al., 2024).

1034

1035 Beam trawl data showed that although there were distinct changes in the estuary fish communities
1036 of the study rivers between 1990 and 2010, little consistent improvement in fish species richness
1037 occurred over that period. It is likely that the estuaries were already clean enough to enable most of
1038 the characteristic species to exist by the early 1990s when quantitative sampling started. It is also
1039 likely that changes in the fish communities over the study period reflected wider changes in North
1040 Sea environmental conditions and fish recruitment as well as local changes in the estuaries (Clark &
1041 Frid, 2001; van der Veer et al., 2022).

1042

1043 Future ecological restoration works on the study catchments should be mainly focused on easing
1044 pollution (with increased emphasis on diffuse pollution), and restoring both physical habitat and
1045 connectivity. In the meantime, it is apparent how crucial reliable quantitative recording and archiving
1046 of fish (and other taxa) community data is across a network of sites (Langford et al., 2012), and this
1047 needs to be a priority for the future. The lack of stream fish community data prior to 1991 that could
1048 be recovered for our study rivers is evidence of the extent of this problem – good quality quantitative
1049 electric fishing records were made in the 1980s, but they were disposed of during the transition from
1050 Northumbrian Water Authority to the privatised water company (Northumbrian Water) and the
1051 creation of the National Rivers Authority. The same is also true for environmental data such as water
1052 quality parameters. We found that many data series available from the EA and its predecessors
1053 were of short duration and suffered problems of changes in location and method that made
1054 assessment of change difficult. For most fish species, accessible quantitative data are available only
1055 since 1991, limiting the quantitative assessment of fish-community restoration success.

1056

1057 **5. Concluding lessons from this study - degradation and recovery of river fish communities** 1058 **across the world**

1059 The primary lessons from this historical study are that declines in fish biodiversity and fisheries caused
1060 by industrialisation, urbanisation and exploitation of rivers occur over long timescales and can be
1061 difficult to reverse, because they are usually multicausal and inherently linked to the socio-economic
1062 fabric of a region. Yet geographical areas under rapid development rarely consider the potential
1063 chronic impacts and limited reversibility of such impacts on fish populations and wider biodiversity,
1064 even when fish provide crucial ecosystem services. For example, in Asia, numerous Chinese rivers
1065 have suffered severe industrial pollution of multiple types in recent decades as economic development
1066 has accelerated, including widespread coal and metal mining impacts (Yin et al., 2025; Yuan et al.,
1067 2020), known to impact fish communities (Daniel et al., 2015). While environmental monitoring in
1068 China's rivers is increasingly extensive, many of these industrially impacted sites do not have good
1069 historical data for fishes, environmental impacts have been poorly managed (Yin et al., 2025) and
1070 restoration actions are at a very early stage of development (Yuan et al., 2020). The lower Ganges
1071 River is suffering the effects of overfishing, flow diversion, and species invasion, resulting in local

1072 extinction of 28 fish species over the past 40 years (Parvez et al., 2023). Recent evidence suggests
1073 that the effects of anthropogenic barriers on Mekong fishes have been considerably underestimated,
1074 as earlier inventories overlooked large numbers of small barriers; basin-scale reassessments indicate
1075 that these structures make major cumulative contributions to connectivity loss, affecting 93% of
1076 assessed fish species in the basin (Sun, Baldan, et al., 2025; Sun et al., 2023).

1077
1078 The strongest lessons from studies like ours are, therefore, of the importance of advocacy and
1079 intervention (Hobbs et al., 2011; Lackey, 2007) to prevent or limit the scale and effects of threats
1080 associated with anthropogenic activities on river fish and fisheries and ecology more widely. This does
1081 not imply an either / or strategy but one that promotes education, legislation and effective planning
1082 that draws a balance between societal and environmental needs, environmental regulation, effective
1083 monitoring and adequate budget provision. For example, protecting priority subcatchments for
1084 threatened species, while allowing development in less sensitive regions is one option for preventing
1085 catchment-wide ecological degradation (Sun, Lucas, et al., 2025; M. Thieme et al., 2023). The high
1086 frequency with which multiple elements of ecologically-sensitive development strategies are missing
1087 in less developed regions of the world, and the resultant effects on threatened freshwater taxa,
1088 indicate there is far to go to institute historical lessons from studies such as ours.

1089

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1641
1642 **Author Contributions Statement**

1643 JS and MCL planned the project. JS and MCL prepared the data. JS and SG performed the
1644 analysis. JS and MCL wrote the original draft. All authors commented and improved the manuscript
1645 and agreed on the final version. JS and MCL supervised this work.

1646
1647 **Financial and Non-Financial Competing Interest statement**

1648 The authors declare no competing interests.

Supplementary information

for

Two centuries of decline and partial recovery of fish in rivers of post-industrial northeast England:
lessons for global rivers in the Anthropocene

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Supplementary Methods

Authorities reports used as data sources for study rivers

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Sampling of fish communities in freshwater reaches of the study rivers

The main sampling method used to quantify community composition and fish abundance in freshwater sections of the Tyne, Wear and Tees has been the use of single or dual anode electric fishing and has been carried out with variable rigour and frequency since the 1980s; unfortunately data from the 1980s was lost and, for most sites only became available and rigorous from the early 1990s under the auspices of the National Rivers Authority. Most of this sampling was aimed at estimating the abundance of juvenile salmonids in stop-netted sections of small streams, but other species captured were enumerated also. Sites were typically sampled on a rolling 3-year cycle, but some site localities changed. Triple-pass fishing was typical, but since the Environment Agency's adoption of the European Committee for Standardization (CEN) standard for electric fishing surveys in Water Framework Directive sampling, single pass surveys have become the norm. In some sections of the lower rivers, especially the Tees, boat-mounted electric fishing was used, and at some sites micromesh seine nets were used to sample young-of-the-year fishes, especially cyprinids. Additional NRA/EA surveying has employed angler census methods.

Supplementary figures

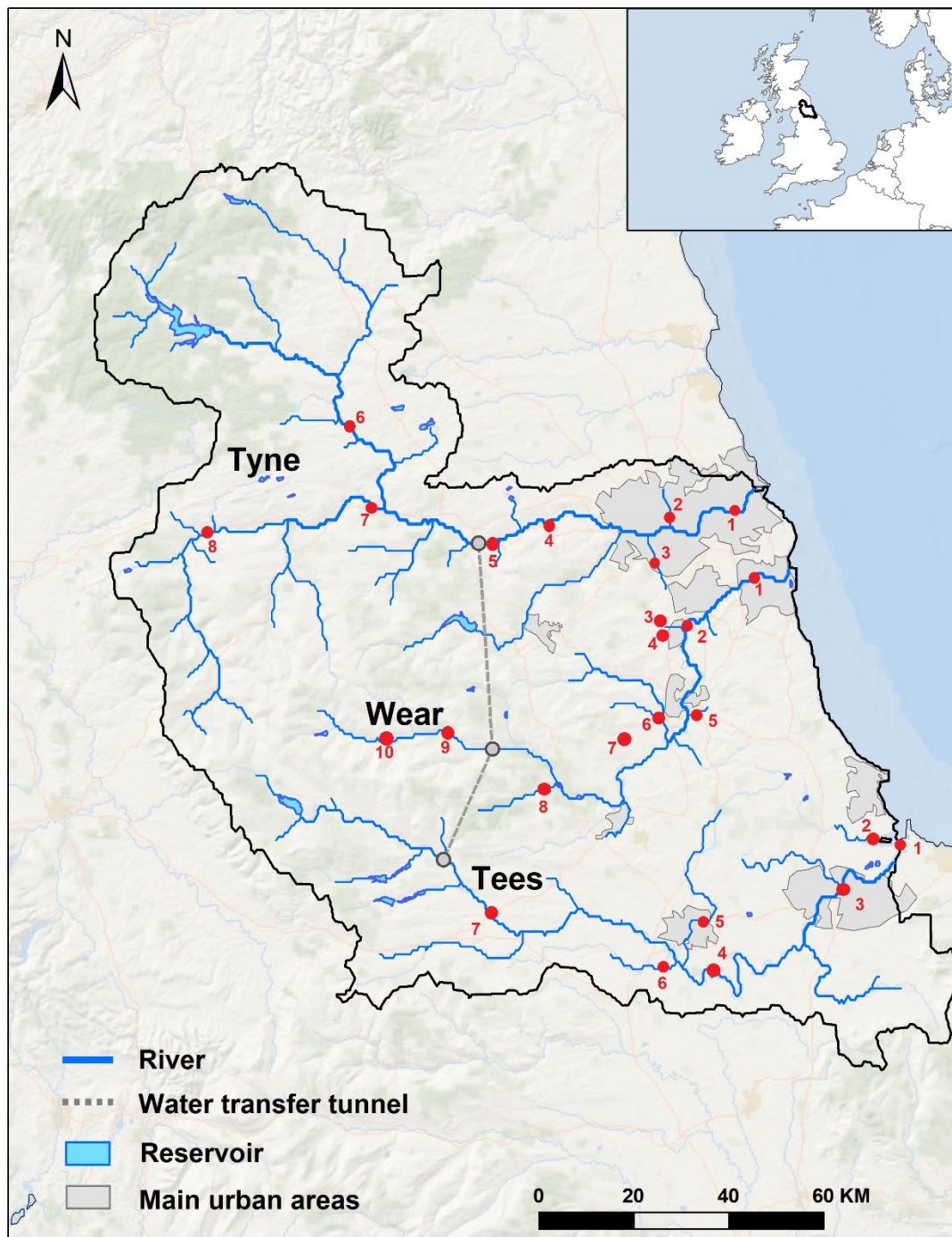


Figure S1. Locations of the Tyne, Wear and Tees river basins, and sites with long-term water chemistry trends for which data are presented in Supplementary Tables.



Figure S2. View of the weir and fish locks (each side of weir) on the River Tees at Dinsdale (18 km upstream of tidal limit) in the early 19th Century, drawn by J.M. Sparks.

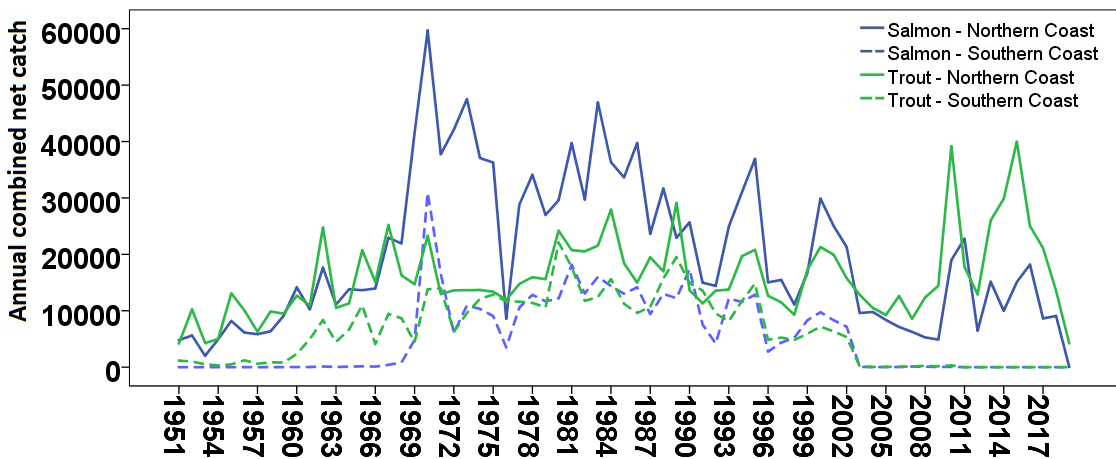


Figure S3. Salmon and sea trout fishery annual net catch (all net types combined), from 1951 to 2019 on northern coast (Tweed to Tyne) and southern coast (Tyne to Tees). Zero catches on southern coast since 2003 are due to the closure of nets. Figure generated using commercial catches reported to government.

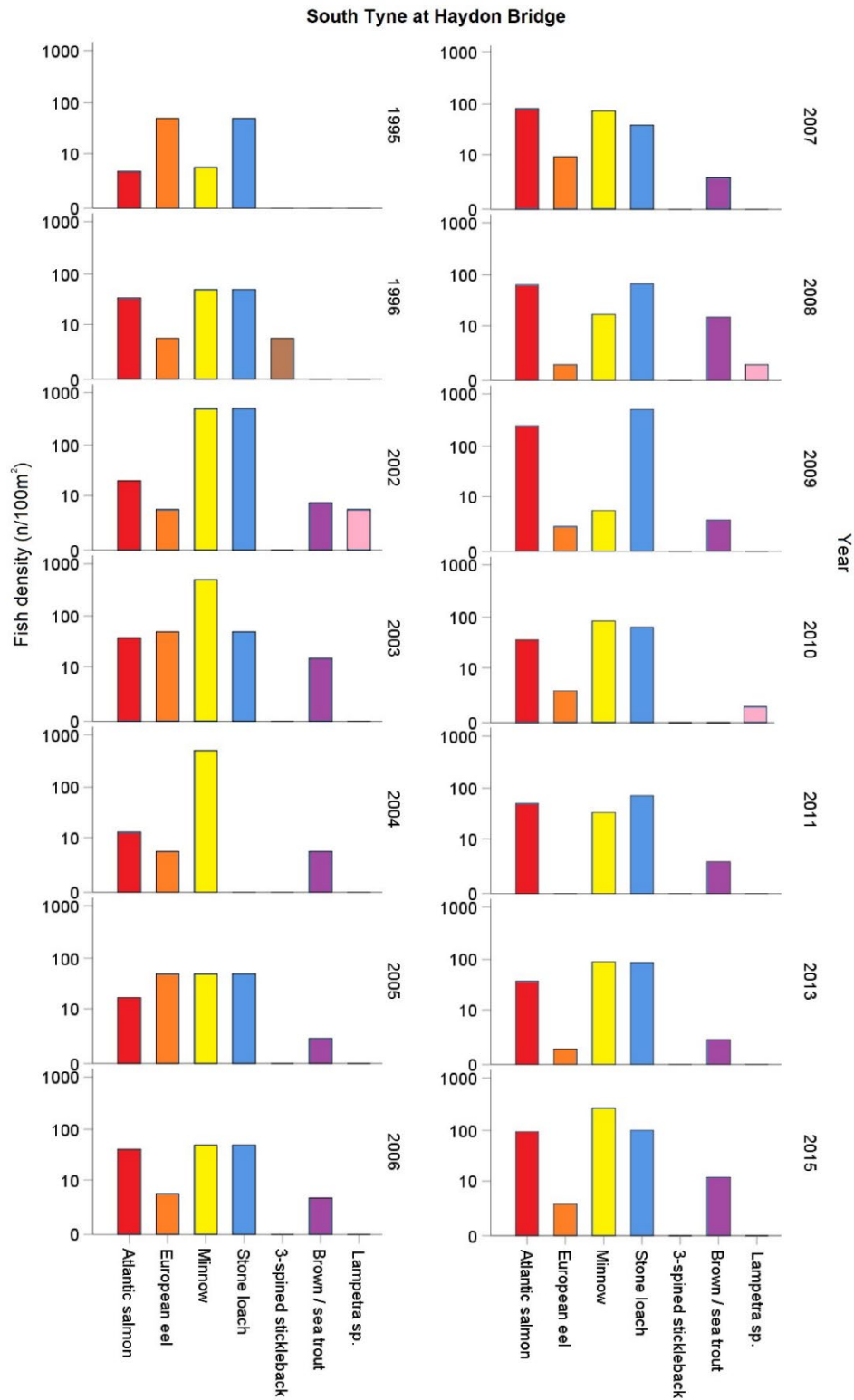


Figure S4. Long-term variation of estimated minimum fish density in the River South Tyne, Haydon Bridge site (upper Tyne catchment) between 1995 and 2015. Fish density was calculated by the single pass electrofishing method.

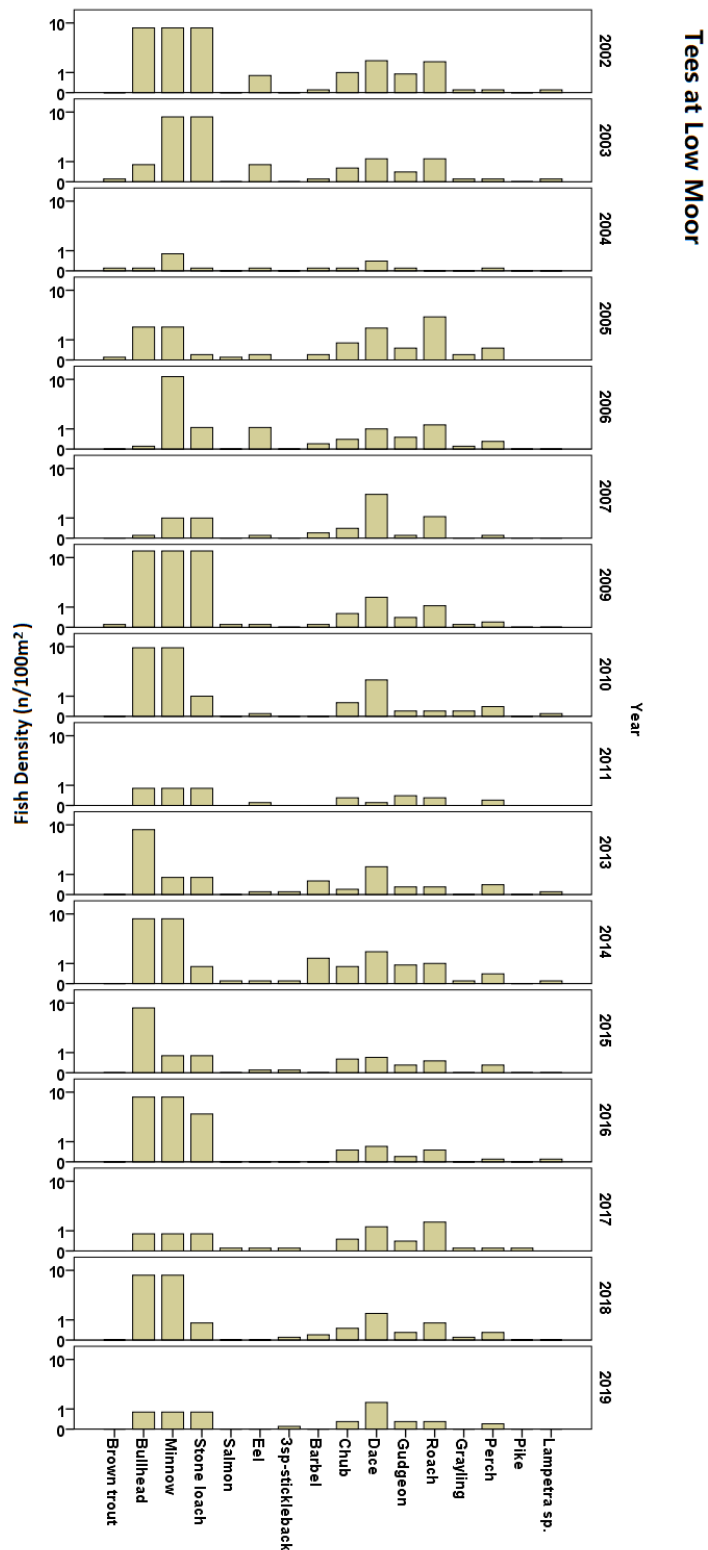


Figure S5. Long-term variation of estimated minimum fish density at River Tees Low Moor site between 2002 and 2019. Fish density was calculated by single pass electrofishing method. Notice: abundance values are on log scales.

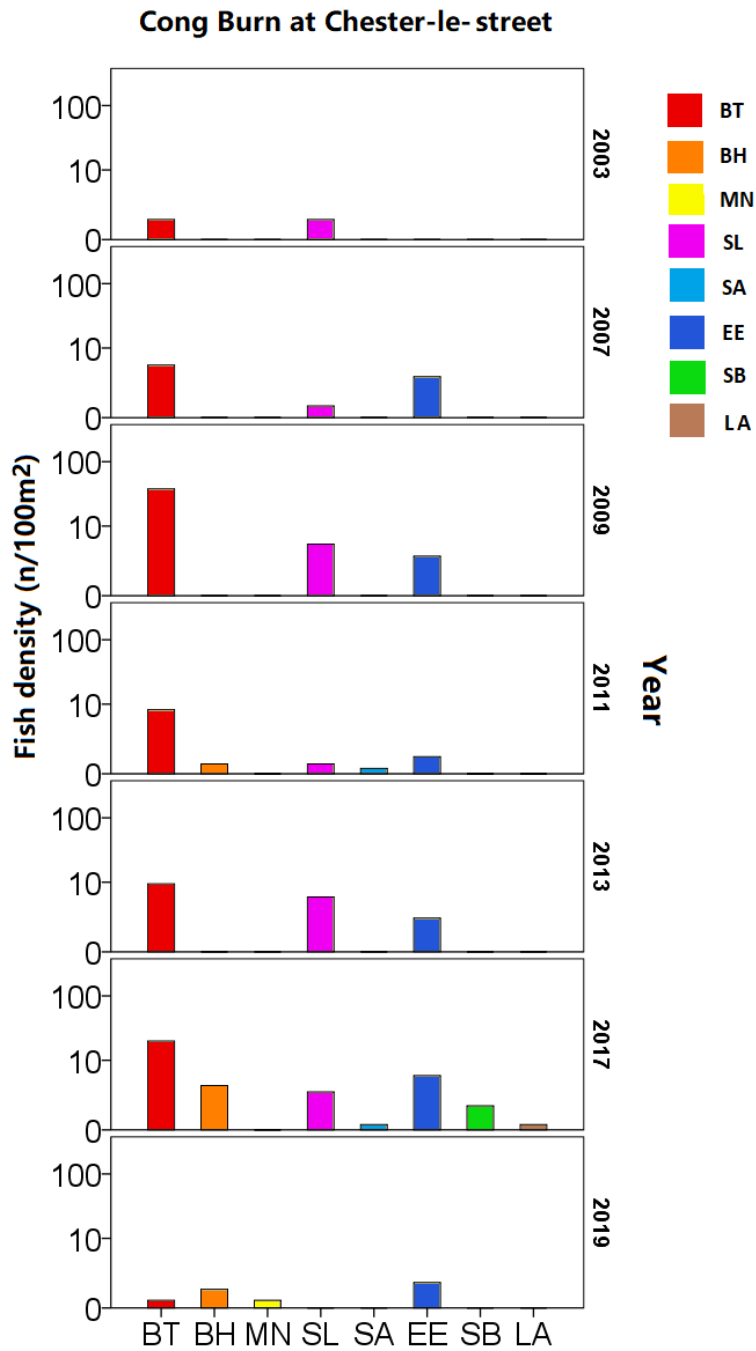


Figure S6 Recolonization of the Cong Burn shown through long-term variation of estimated minimum fish density (2003-2017: single pass electric fishing; 2019: first run data from three pass electric fishing; estimated three pass capture efficiency: trout 63.6%, bullhead 84.6%, minnow 100%) at Cong Burn. BT: brown trout, BH: bullhead, MN: minnow SL: stone loach, SA: Atlantic salmon, EE: eel, SB: three-spined stickleback, LA: *Lampetra* sp. Note the log scale.

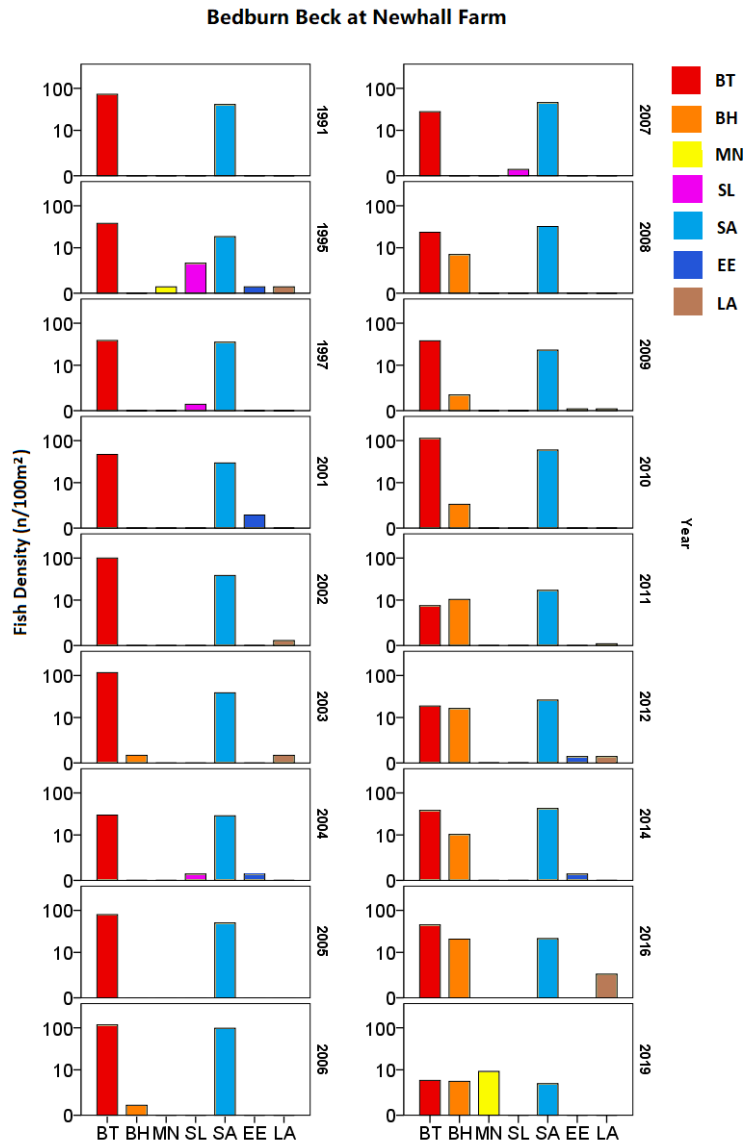


Figure S7. Long-term variation of estimated minimum fish density (1991-1997, 2007- 2016: single pass electric fishing; 2001-2006, 2019: first run data from three pass electric fishing; estimated three pass capture efficiency: trout 93.7%, salmon 84.2%, bullhead 60.6%, minnow 77.0%) at Bedburn Beck Newhall Farm site (upper reaches of River Wear). BT: brown trout, SA: Atlantic salmon, BH: bullhead, LA: *Lampetra* sp., MN: minnow, SL: stone loach, EE: eel. Notice: the 1991 survey only counted trout and salmon. Note the log scale.

Table S1. Freshwater fish species in the river channels of the Tyne, Wear and Tees, and their indicative contemporary abundance, mostly based on data sources given in Sun (2021). Abundance status is relative only (based on electric fishing surveys, net surveys and angler catches) and refers to abundance within the freshwater channels of the rivers, rather than stillwaters within the catchment. N= Native, I = Introduced (including translocated from other catchments within Great Britain), S = stocked. The Tees, Wear and Tyne occur in the south-north transition zone from warmer water (e.g. cyprinid) species to cooler water (e.g. salmonid species). The Tees (southern-most) is adjacent to the Humber river basin which has relatively wider freshwater fish species diversity (Wheeler, 1969) and some of those species recorded in this table for the Tees as 'native' may have arrived from the Humber system and now be naturalised. It is likely that a proportion of these species have been introduced to the more northerly Wear and Tyne (listed as I / I?), but details are poor. The Tees contains a greater proportion of lowland river habitat suited to cyprinid, esocid and percid fish than do the Wear and Tyne.

Order	Species	Tyne - Status	Tyne - Abundance	Wear - Status	Wear - Abundance	Tees - Status	Tees - Abundance
Petromyzontiformes	<i>Petromyzon marinus</i>	N	+	N	+	N	+
Petromyzontiformes	<i>Lampetra fluviatilis</i>	N	++	N	++	N	++
Petromyzontiformes	<i>Lampetra planeri</i>	N	+++	N	+++	N	+++
Anguilliformes	<i>Anguilla anguilla</i>	N	++	N	++	N	++
Cypriniformes	<i>Abramis brama</i>	I	+	I	+	N?	++
Cypriniformes	<i>Barbatula barbatula</i>	N	+++	N	+++	N	+++
Cypriniformes	<i>Barbus barbus</i>			I-S	+	N?	+
Cypriniformes	<i>Cyprinus carpio</i>			I	+	I	+
Cypriniformes	<i>Gobio gobio</i>	N?	+	N	+	N	++
Cypriniformes	<i>Leuciscus leuciscus</i>	I?	++	I?		N	+++
Cypriniformes	<i>Leuciscus idus</i>					I	+
Cypriniformes	<i>Phoxinus phoxinus</i>	N	+++	N	+++	N	+++
Cypriniformes	<i>Rutilus rutilus</i>	I?	++	I?	++	N	+++
Cypriniformes	<i>Squalius cephalus</i>	I?	+	I?	++	N	++
Cypriniformes	<i>Scardinius erythrophthalmus</i>					N	+

Cypriniformes	<i>Tinca tinca</i>					N? -S	+
Esociformes	<i>Esox lucius</i>	I?	+	I?	+	N	+
Osmeriformes	<i>Osmerus eperlanus</i>					N	Extirpated
Salmoniformes	<i>Oncorhynchus mykiss</i>	I-S	+	I-S	+	I-S	+
Salmoniformes	<i>Salmo salar</i>	N	+++	N	++	N	+
Salmoniformes	<i>Salmo trutta</i>	N	+++	N	+++	N	+++
Salmoniformes	<i>Thymallus thymallus</i>	I?	+	I?	++	N	++
Gasterosteiformes	<i>Gasterosteus aculeatus</i>	N	++	N	++	N	++
Gasterosteiformes	<i>Pungitius pungitius</i>	N	+	N	+	N	+
Scorpaeniformes	<i>Cottus gobio</i> *	N	+++	N	+++	N	+++
Perciformes	<i>Liza ramada</i>	N	+	N	+	N	+
Perciformes	<i>Perca fluviatilis</i>	N	+	N	++	N	++
Perciformes	<i>Pomatoschistus microps</i>	N	+	N	+	N	+
Pleuronectiformes	<i>Platichthys flesus</i>	N	+	N	+	N	+

*Cottus gobio** - normally recorded as this species in Great Britain, but Freyhof *et al* (2005) indicate this to be *C. perifretum* based on the observed distribution ranges and molecular genetics of these species across Europe.

Table S3. Linear model summaries of changes in key water quality parameters in the Tyne catchment (S1-S8) since the 1970s (date ranges reflect availability of data for analysis). Site numbers increase from downstream to upstream, with lowest site numbers are nearest to the sea. Locations of sites referred to are shown in Fig. S1.

Site	Parameter	Periods	<i>df</i>	<i>t</i>	<i>P</i>
1 Tidal reach	DO	1991-2020	1,122	2.49	0.014
	Ammonia	1989-2002	1,128	-2.48	0.014
	Orthophosphate	1991-2013	1,21	-2.37	0.027
	Zinc	1989-2002	1,72	-1.84	0.070
	Lead	1989-2002	1,70	-0.08	0.934
2 Ouse Burn	DO	1973-2006	1,347	-2.38	0.018
	BOD	1973-2006	1,380	-2.50	0.013
	Ammonia	1973-2006	1,370	-2.65	0.008
	Nitrate	1973-2006	1,359	-1.96	0.051
	Orthophosphate	1978-2006	1,259	2.25	0.025
	Zinc	1974-2006	1,347	-4.42	<0.001
	Lead	1976-2002	1,123	-5.24	<0.001
3 River Team	DO	1973-2019	1,494	4.03	<0.001
	BOD	1973-2013	1,488	-10.59	<0.001
	Ammonia	1973-2019	1,536	-13.61	<0.001
	Nitrate	1973-2019	1,527	-0.47	0.639
	Phosphorus	1994-2019	1,276	-11.60	<0.001
	Zinc	1973-2019	1,517	-36.08	<0.001
	Lead	1973-2019	1,503	-21.89	<0.001
4 Wylam Bridge	DO	1974-2020	1,564	2.18	0.030
	BOD	1974-2014	1,547	-5.16	<0.001
	Ammonia	1974-2020	1,582	-12.02	<0.001
	Nitrate	1974-2020	1,556	-2.94	0.003
	Phosphorus	1974-2020	1,441	-6.43	<0.001
	Zinc	1974-2020	1,480	-4.19	<0.001
	Lead	1974-2020	1,474	-11.11	<0.001
5 Bywell	DO	1973-2019	1,457	0.41	0.684
	BOD	1973-2007	1,412	-3.58	0.000
	Ammonia	1973-2019	1,563	-4.68	<0.001
	Nitrate	1973-2019	1,455	0.91	0.362
	Orthophosphate	1974-2019	1,451	-3.87	0.000
	Zinc	1974-2019	1,454	-2.43	0.016
	Lead	1973-2019	1,203	-6.02	<0.001
6 Wark	DO	1974-2006	1,280	1.23	0.221
	BOD	1974-2006	1,353	-0.86	0.391
	Ammonia	1974-2006	1,354	-1.01	0.312
	Nitrate	1974-2006	1,249	1.72	0.087
	Orthophosphate	1974-2006	1,231	0.74	0.458

	Zinc	1974-2006	1,273	-9.40	<0.001
	Lead	1974-1994	1,118	-13.25	<0.001
7	DO	1973-2020	1,540	-1.50	0.135
Warden Bridge	BOD	1973-2014	1,482	0.14	0.889
	Ammonia	1973-2020	1,557	-2.21	0.028
	Nitrate	1973-2020	1,517	-0.67	0.504
	Phosphorus	1974-2020	1,441	-5.78	<0.001
	Zinc	1974-2019	1,415	-1.81	0.070
	Lead	1974-2019	1,408	-4.88	<0.001
8	DO	1979-2020	1,415	0.80	0.427
Haltwhistle	BOD	1979-2007	1,338	-1.55	0.123
	Ammonia	1979-2020	1,436	-2.27	0.024
	Nitrate	1979-2020	1,339	-2.61	0.009
	Orthophosphate	1979-2020	1,243	-4.71	<0.001
	Zinc	1979-2013	1,347	-4.15	<0.001
	Lead	1979-2002	1,206	-4.79	<0.001

Table S4. Issues in the Tyne catchment preventing waters reaching good ecological status by cycle 2 (2015-2021) of the Water Framework Directive and the sectors identified as contributing to them (the numbers in the table are counts of the reasons for not achieving good status in water bodies).

	Agriculture and rural land management	Domestic General Public	Industry	Local and Central Government	Mining and quarrying	Urban and transport	Water Industry	Other	Sector under investigation	Total
Changes to the natural flow and levels of water	-	-	-	-	-	-	2	-	-	2
Pollution from rural areas	28	-	-	-	-	-	-	-	-	28
Pollution from abandoned mines	-	-	-	-	52	-	-	-	-	52
Pollution from waste water	-	-	-	-	-	-	9	-	-	9
Physical modifications	1	1	2	4	-	14	29	2	6	59
Pollution from towns, cities and transport	-	-	1	-	-	6	11	-	-	18
Non-native invasive species	-	-	-	-	-	-	-	-	-	0

Table S5 Linear model summaries of changes in key water quality parameters in the Wear catchment (S1-S10) since the 1970s (date ranges reflect availability of data for analysis). Site numbers increase from downstream to upstream, with lowest site numbers are nearest to the sea. Locations of sites referred to are shown in Fig. S2.

Site	Parameter	Periods	<i>df</i>	<i>t</i>	<i>P</i>
1 Tidal reach	DO	1989-2020	1,62	2.64	0.011
	Ammonia	1988-2002	1,115	-3.54	<0.001
	Orthophosphate	1989-2020	1,27	-3.09	0.005
	Zinc	1989-2002	1,45	-1.71	0.094
	Lead	1989-2002	1,43	0.03	0.974
2 Lambton Bridge	DO	1973-2019	1,515	3.13	0.002
	BOD	1973-2014	1,539	-8.35	<0.001
	Ammonia	1973-2019	1,591	-11.81	<0.001
	Nitrate	1973-2019	1,537	2.00	0.046
	Phosphorus	1974-2019	1,389	-11.85	<0.001
	Zinc	1974-2019	1,427	-5.84	<0.001
3 Twizell Burn	DO	1979-2019	1,270	4.15	<0.001
	BOD	1979-2006	1,81	-1.98	0.050
	Ammonia	1979-2019	1,287	-4.12	<0.001
	Nitrate	1979-2019	1,283	2.98	0.003
	Orthophosphate	1995-2019	1,232	-19.49	<0.001
	Zinc	1989-2011	1,108	-2.21	0.029
4 Cong Burn	DO	1979-2019	1,205	1.13	0.262
	BOD	1979-2006	1,178	-3.46	<0.001
	Ammonia	1979-2019	1,221	-3.32	<0.001
	Nitrate	1979-2019	1,200	-2.01	0.046
	Orthophosphate	1995-2019	1,167	-2.64	0.009
5 Old Durham Beck	Zinc	1995-2004	1,101	-1.73	0.087
	DO	1973-2019	1,448	-0.30	0.763
	BOD	1973-2015	1,393	-2.41	0.016
	Ammonia	1973-2019	1,496	-4.64	<0.001
	Nitrate	1973-2019	1,484	6.73	<0.001
	Orthophosphate	1977-2019	1,401	4.03	<0.001
6 Deerness	Zinc	1974-2006	1,158	-5.43	<0.001
	Lead	1979-2002	1,106	-3.71	<0.001
	DO	1973-2019	1,494	-0.36	0.721
	BOD	1973-2007	1,408	-2.62	0.009
	Ammonia	1973-2019	1,510	-6.30	<0.001
7 Brancepeth Beck	Nitrate	1973-2019	1,460	5.28	<0.001
	Zinc	1974-2013	1,260	-4.67	<0.001
	Lead	1994-2002	1,103	-3.50	<0.001
	DO	1991-2004	1,123	1.40	0.164
	BOD	1991-2004	1,146	1.76	0.080

	Ammonia	1991-2004	1,145	-2.78	0.006
	Nitrate	1991-2004	1,110	0.23	0.822
	Orthophosphate	1995-2004	1,95	-0.23	0.820
	Zinc	1995-2004	1,100	-1.18	0.243
8	DO	1991-2019	1,248	1.73	0.085
Bedburn Beck	BOD	1991-2016	1,226	-0.82	0.413
	Ammonia	1991-2019	1,283	-7.23	<0.001
	Nitrate	1991-2019	1,207	-4.96	<0.001
	Orthophosphate	1992-2019	1,259	-5.66	<0.001
	Zinc	1992-2019	1,252	-2.39	0.018
9	DO	1973-2012	1,422	-0.28	0.780
Stanhope	BOD	1973-2007	1,382	-1.34	0.183
	Ammonia	1973-2012	1,446	-2.98	0.003
	Nitrate	1973-2012	1,392	-1.99	0.048
	Orthophosphate	1974-2012	1,329	-1.39	0.166
	Zinc	1974-2011	1,384	-7.35	<0.001
	Lead	1974-2011	1,295	-3.05	0.003
10	DO	1976-2019	1,262	2.10	0.037
Rookhope Burn	BOD	1976-2007	1,262	0.74	0.459
	Ammonia	1976-2019	1,307	-2.84	0.005
	Nitrate	1976-2019	1,232	-2.24	0.026
	Orthophosphate	1978-2019	1,175	-2.90	0.004
	Zinc	1978-2019	1,263	-12.34	<0.001
	Lead	1978-2019	1,184	-4.36	<0.001

Table S6. Issues in the Wear catchment preventing waters reaching good ecological status under WFD by cycle 2 (2015-2021) and the sectors identified as contributing to them (the numbers in the table are counts of the reasons for not achieving good status in water bodies).

	Agriculture and rural land management	Domestic General Public	Industry	Local and Central Gov	Mining and quarrying	Recreat -ion	Urban and transport	Water Industry	Other	No sector responsible	Sector under investi- gation	Total
Changes to the natural flow and levels of water	-	-	-	-	-	-	-	2	-	-	-	2
Pollution from rural areas	22	-	-	-	-	-	-	-	-	-	-	22
Pollution from abandoned mines	-	-	-	-	34	-	-	-	-	-	-	34
Pollution from waste water	-	-	1	-	-	-	-	55	2	-	-	58
Physical modifications	8	-	-	9	-	2	17	8	1	-	3	48
Pollution from towns, cities and transport	-	1	1	-	-	-	2	30	-	-	-	34
Non-native invasive species	-	-	-	-	-	-	-	-	-	8	-	8

Table S7 Linear model summaries of changes in key water quality parameters in the Tees catchment (S1-S7) since the 1970s (date ranges reflect availability of data for analysis). Site numbers increase from downstream to upstream, with lowest site numbers are nearest to the sea. Locations of sites referred to are shown in Fig. S2.

Site	Parameter	Periods	<i>df</i>	<i>t</i>	<i>P</i>
1 Estuary	DO	1993-2019	1,203	-2.42	0.016
	Ammonia	1990-2012	1,269	-9.46	<0.001
	Nitrate	1992-2019	1,216	-2.38	0.018
	Orthophosphate	1992-2019	1,330	-3.40	<0.001
	Zinc	1989-2013	1,181	-5.46	<0.001
	Lead	1989-2011	1,173	-2.77	0.006
2 Greatham Creek	DO	1997-2002	1,59	1.31	0.194
	BOD	1984-2002	1,78	-3.01	0.004
	Ammonia	1984-2002	1,79	-4.37	<0.001
	Nitrate	1984-2002	1,65	-4.80	<0.001
	Orthophosphate	1997-2002	1,89	-1.27	0.206
	Zinc	1984-2002	1,68	-4.03	<0.001
	Lead	1984-2002	1,68	-9.32	<0.001
3 Newport Bridge	DO	1983-2020	1,252	6.93	<0.001
	Ammonia	1983-2012	1,361	-14.46	<0.001
	Nitrate	1983-2020	1,322	-1.52	0.130
4 Low Hail	DO	1973-2013	1,410	-1.36	0.175
	BOD	1973-2008	1,391	-5.52	<0.001
	Ammonia	1973-2013	1,446	-7.56	<0.001
	Nitrate	1973-2012	1,421	-4.15	<0.001
	Orthophosphate	1978-2012	1,267	-3.47	<0.001
	Zinc	1974-2013	1,252	-7.41	<0.001
	Lead	1978-2003	1,101	-3.31	0.001
5 River Skerne	DO	1973-2019	1,438	8.48	<0.001
	BOD	1973-2008	1,400	-14.71	<0.001
	Ammonia	1973-2019	1,452	-21.07	<0.001
	Nitrate	1973-2019	1,448	-3.77	<0.001
	Orthophosphate	1978-2019	1,311	-7.85	<0.001
	Zinc	1974-2004	1,179	-6.74	<0.001
	Lead	1989-2003	1,145	-2.42	0.017
6 Clow Beck	DO	1976-2017	1,306	-0.62	0.539
	BOD	1976-2013	1,338	-8.36	<0.001
	Ammonia	1976-2017	1,377	-8.96	<0.001
	Nitrate	1976-2017	1,377	-4.10	<0.001
	Orthophosphate	1978-2017	1,276	-7.64	<0.001
	Zinc	1989-2013	1,247	-2.39	0.017
	Lead	1989-2003	1,102	-0.82	0.417
7 Egglestone Abbey	DO	1984-2019	1,242	1.44	0.152
	BOD	1984-2006	1,225	-2.04	0.043

Ammonia	1984-2019	1,308	-5.71	<0.001
Nitrate	1984-2019	1,236	-3.38	<0.001
Orthophosphate	1995-2019	1,244	-4.62	<0.001
Zinc	1992-2019	1,231	-4.23	<0.001
Lead	1992-2019	1,185	-3.54	<0.001

Table S8. Issues in the Tees catchment currently preventing waters reaching good status (second WFD cycle, 2015-2021) and the sectors identified as contributing to them (the numbers in the table are counts of the reasons for not achieving good status in water bodies).

	Agriculture and rural land management	Domestic General Public	Local and Central Gov	Mining and quarrying	Navi-gation	Urban and transport	Water Industry	Other	No sector responsible	Sector under investi-gation	Total
Changes to the natural flow and levels of water	1	-	-	-	-	-	7	-	-	1	9
Pollution from rural areas	73	-	-	-	-	-	-	-	-	-	73
Pollution from abandoned mines	-	-	-	11	-	-	-	-	-	-	11
Pollution from waste water	-	2	-	-	-	-	55	3	-	-	60
Physical modifications	18	1	24	-	15	17	8	3	-	3	89
Pollution from towns, cities and transport	-	6	1	-	-	6	34	-	-	-	47
Non-native invasive species	-	-	-	-	-	-	-	-	1	-	1

Table S9. Results of the SIMPER (Similarity Percentage) analysis showing response of fish species to riverine habitat in 1980s or 2000s in the River Tyne and their contribution to the overall variation, based on Bray–Curtis dissimilarity index.

Species	Average dissimilarity	sd	ratio	1980s	2000s	p-value	Cumulative Contribution (%)	Contribution (%)
<i>Clupea.harengus</i>	0.11949	0.05595	2.1355	2.64	0.064	0.001	24.7	24.7
<i>Merlangius.merlangus</i>	0.05703	0.0413	1.381	2.1354	1.7035	0.325	36.5	11.8
<i>Sprattus.sprattus</i>	0.04075	0.02543	1.6021	0.9675	0.2522	0.002	44.9	8.4
<i>Platichthys.flesus</i>	0.03981	0.02723	1.4619	1.8592	2.205	0.542	53.1	8.2
<i>Pollachius.virens</i>	0.03846	0.04915	0.7826	0.6573	0.8206	0.459	61.1	8
<i>Pleuronectes.platessa</i>	0.03567	0.03501	1.0188	0.7403	1.0799	0.654	68.5	7.4
<i>Agonus.cataphractus</i>	0.02168	0.0186	1.1653	0.2517	0.6535	0.267	73	4.5
<i>Gadus.morhua</i>	0.02083	0.0174	1.1972	0.6112	0.8373	0.938	77.3	4.3
<i>Limanda.limanda</i>	0.01377	0.0107	1.2862	0.3842	0.3262	0.721	80.1	2.8
<i>Pomatoschistus.minutus</i>	0.01309	0.01073	1.2202	0.2356	0.2884	0.581	82.8	2.7
<i>Zoarces.viviparus</i>	0.01294	0.01161	1.1137	0.5635	0.4627	0.893	85.5	2.7
<i>Myoxocephalus.scorpius</i>	0.01182	0.00879	1.3447	0.1765	0.2626	0.379	87.9	2.4
<i>Salmo.trutta</i>	0.00826	0.00518	1.5959	0.172	0	0.01	89.6	1.7
<i>Anguilla.anguilla</i>	0.00798	0.00487	1.6366	0.1775	0.1085	0.915	91.3	1.7
<i>Pollachius.pollachius</i>	0.00686	0.01915	0.358	0	0.1884	0.657	92.7	1.4
<i>Taurulus.bubalis</i>	0.0054	0.00886	0.6095	0.0116	0.1214	0.682	93.8	1.1
<i>Ciliata.mustela</i>	0.00421	0.00301	1.3988	0.0673	0.0863	0.566	94.7	0.9
<i>Solea.solea</i>	0.00245	0.00242	1.0128	0.0379	0.0372	0.735	95.2	0.5
<i>Gasterosteus.aculeatus</i>	0.00241	0.00202	1.1929	0.051	0	0.004	95.7	0.5
<i>Eutrigla.gurnardus</i>	0.00238	0.00238	0.9999	0.0518	0	0.012	96.2	0.5
<i>Entelurus.aequoreus</i>	0.00225	0.00471	0.4778	0	0.0394	0.5	96.7	0.5

<i>Liparis.montagui</i>	0.00188	0.00395	0.4759	0	0.0379	0.488	97.1	0.4
<i>Hyperoplus.lanceolatus</i>	0.0014	0.00424	0.3303	0	0.0236	0.73	97.4	0.3
<i>Liparis.liparis</i>	0.00121	0.00133	0.9124	0.026	0	0.756	97.6	0.2
<i>Gobiidae</i>	0.00118	0.00358	0.3306	0	0.025	0.722	97.8	0.2
<i>Syngnathus.rostellatus</i>	0.00117	0.0015	0.7831	0.0181	0.0102	0.946	98.1	0.3
<i>Echiichthys.vipera</i>	0.00112	0.00188	0.5959	0.005	0.0193	0.401	98.3	0.2
<i>Leuciscus.leuciscus</i>	0.00102	0.00189	0.5395	0.0222	0	0.729	98.5	0.2
<i>Lampetra.fluviatilis</i>	0.00099	0.00157	0.6316	0.0164	0.0072	0.63	98.7	0.2
<i>Petromyzontidae</i>	0.00097	0.00292	0.3306	0	0.0204	0.722	98.9	0.2
<i>Pleuronectidae</i>	0.00097	0.00292	0.3306	0	0.0204	0.722	99.1	0.2
<i>Syngnathus.acus</i>	0.00091	0.00274	0.3307	0	0.0209	0.73	99.3	0.2
<i>Ammodytidae</i>	0.00068	0.00147	0.4652	0.005	0.0102	0.537	99.5	0.2
<i>Salmo.salar</i>	0.00059	0.0012	0.489	0.0065	0.0072	0.983	99.6	0.1
<i>Cyclopterus.lumpus</i>	0.00049	0.00148	0.3305	0	0.0091	0.785	99.7	0.1
<i>Pholis.gunnellus</i>	0.00033	0.00089	0.3717	0.008	0	0.944	99.8	0.1
<i>Melanogrammus.aeglefinus</i>	0.00033	0.00089	0.3717	0.008	0	0.116	99.8	0
<i>Rutilus.rutilus</i>	0.00029	0.00079	0.3705	0.0061	0	0.756	99.9	0.1
<i>Lophius.piscatorius</i>	0.00029	0.00078	0.3712	0.0065	0	0.103	99.9	0
<i>Callionymus.lyra</i>	0.00029	0.00077	0.3704	0.0059	0	0.605	100	0.1

Table S10. Results of the SIMPER (Similarity Percentage) analysis showing response of fish species to riverine habitat in 1980s or 2000s in the River Wear and their contribution to the overall variation, based on Bray–Curtis dissimilarity index.

Species	Average dissimilarity	sd	ratio	1980s	2000s	<i>p</i> -value	Cumulative contribution (%)	Contribution (%)
<i>Platichthys.flesus</i>	0.16482	0.09346	1.7635	4.284	1.8764	0.104	25.9	25.9
<i>Clupea.harengus</i>	0.07941	0.0421	1.8865	1.396	0.0953	0.002	38.4	12.5
<i>Merlangius.merlangus</i>	0.05948	0.04963	1.1985	0.977	0.7611	0.405	47.8	9.4
<i>Sprattus.sprattus</i>	0.04523	0.06878	0.6575	0.859	0.1584	0.379	54.9	7.1
<i>Zoarcetes.viviparus</i>	0.04521	0.02301	1.9651	0.841	0.1324	0.003	62	7.1
<i>Pleuronectes.platessa</i>	0.04347	0.03258	1.3343	0.82	0.6966	0.871	68.8	6.8
<i>Pollachius.virens</i>	0.04069	0.07179	0.5667	0.752	0.0369	0.375	75.2	6.4
<i>Agonus.cataphractus</i>	0.02891	0.01782	1.6222	0	0.4663	0.006	79.8	4.6
<i>Limanda.limanda</i>	0.02835	0.02288	1.239	0	0.4527	0.015	84.2	4.4
<i>Gadus.morhua</i>	0.02583	0.01586	1.6291	0.022	0.424	0.009	88.3	4.1
<i>Pomatoschistus.minutus</i>	0.0204	0.01471	1.3869	0.163	0.3981	0.389	91.5	3.2
<i>Myoxocephalus.scorpius</i>	0.0097	0.01278	0.7585	0	0.1488	0.217	93	1.5
<i>Anguilla.anguilla</i>	0.00897	0.00793	1.1315	0.13	0.0891	0.918	94.5	1.5
<i>Gasterosteus.aculeatus</i>	0.00673	0.00841	0.7998	0.089	0.0476	0.381	95.5	1
<i>Syngnathus.rostellatus</i>	0.00506	0.01139	0.444	0.066	0.0149	0.274	96.3	0.8
<i>Salmo.trutta</i>	0.00463	0.00824	0.5618	0.088	0	0.457	97	0.7
<i>Pollachius.pollachius</i>	0.00364	0.00688	0.5298	0	0.0601	0.563	97.6	0.6
<i>Solea.solea</i>	0.00344	0.00561	0.6133	0	0.057	0.284	98.1	0.5
<i>Entelurus.aequoreus</i>	0.00324	0.00814	0.3982	0	0.0476	0.572	98.7	0.6
<i>Ammodytes.tobianus</i>	0.00219	0.00549	0.3999	0	0.0382	0.689	99	0.3
<i>Salmo.salar</i>	0.00137	0.00343	0.3993	0	0.0224	0.558	99.2	0.2
<i>Liparis.liparis</i>	0.00127	0.00318	0.3995	0	0.0211	0.886	99.4	0.2
<i>Lampetra.fluviatilis</i>	0.00097	0.00243	0.3993	0	0.0159	0.609	99.6	0.2
<i>Taurulus.bubalis</i>	0.00097	0.00243	0.3993	0	0.0159	0.767	99.7	0.1
<i>Ciliata.mustela</i>	0.0009	0.00225	0.3995	0	0.0149	0.885	99.9	0.2
<i>Myxine.glutinosa</i>	0.0009	0.00225	0.3995	0	0.0149	0.557	100	0.1

Table S11. Results of the SIMPER (Similarity Percentage) analysis showing response of fish species to riverine habitat in 1980s or 2000s in the River Tees and their contribution to the overall variation, based on Bray–Curtis dissimilarity index.

Species	Average dissimilarity	sd	ratio	1980s	2000s	p-value	Cumulative contribution (%)	Contribution (%)
<i>Pleuronectes.platessa</i>	0.12224	0.0768	1.5916	0.7801	2.5851	0.013	19.5	19.5
<i>Merlangius.merlangus</i>	0.05999	0.04495	1.3346	0.716	1.543	0.328	29.1	9.6
<i>Limanda.limanda</i>	0.05675	0.04461	1.2723	0.3869	1.2863	0.027	38.1	9
<i>Agonus.cataphractus</i>	0.05245	0.04236	1.2381	0.0564	0.8528	0.001	46.5	8.4
<i>Gadus.morhua</i>	0.0452	0.03252	1.3902	0.2795	1.0238	0.041	53.7	7.2
<i>Sprattus.sprattus</i>	0.04462	0.05098	0.8753	0.4555	0.5586	0.872	60.8	7.1
<i>Platichthys.flesus</i>	0.04045	0.0335	1.2075	0.9155	0.7522	0.796	67.2	6.4
<i>Pollachius.virens</i>	0.02851	0.0403	0.7075	0.2764	0.3602	0.481	71.8	4.6
<i>Pomatoschistus.minutus</i>	0.02054	0.01611	1.2749	0.2865	0.4095	0.626	75.1	3.3
<i>Gasterosteus.aculeatus</i>	0.01946	0.02437	0.7987	0.3175	0	0.437	78.2	3.1
<i>Clupea.harengus</i>	0.0145	0.01295	1.12	0.2382	0.0565	0.917	80.5	2.3
<i>Zoarces.viviparus</i>	0.01442	0.01125	1.282	0.1813	0.2459	0.614	82.8	2.3
<i>Myoxocephalus.scorpius</i>	0.01355	0.01823	0.7432	0.0529	0.2022	0.302	85	2.2
<i>Taurulus.bubalis</i>	0.01229	0.02042	0.6017	0.0303	0.1889	0.045	86.9	1.9
<i>Echiichthys.vipera</i>	0.01077	0.01148	0.9389	0	0.1809	0.011	88.6	1.7
<i>Anguilla.anguilla</i>	0.00894	0.01115	0.802	0.1325	0.0379	0.892	90.1	1.5
<i>Solea.solea</i>	0.00813	0.00652	1.2473	0	0.133	0.001	91.4	1.3
<i>Ciliata.mustela</i>	0.00746	0.00682	1.0934	0	0.1179	0.042	92.5	1.1
<i>Pholis.gunnellus</i>	0.00606	0.01024	0.5924	0.0157	0.0868	0.075	93.5	1
<i>Callionymus.lyra</i>	0.00528	0.00545	0.9688	0.0564	0.0657	0.767	94.4	0.9
<i>Entelurus.aequoreus</i>	0.0052	0.00968	0.5372	0	0.0737	0.007	95.2	0.8
<i>Eutrigla.gurnardus</i>	0.00464	0.00683	0.6789	0.0708	0	0.471	95.9	0.7
<i>Trisopterus.luscus</i>	0.00318	0.00616	0.516	0	0.0463	0.022	96.4	0.5

<i>Aspitrigla cuculus</i>	0.00268	0.00629	0.4263	0	0.0444	0.667	96.9	0.5
<i>Abramis brama</i>	0.00257	0.00751	0.3422	0.0321	0	0.648	97.3	0.4
<i>Dicentrarchus labrax</i>	0.00232	0.00476	0.4869	0	0.0396	0.018	97.6	0.3
<i>Melanogrammus aeglefinus</i>	0.0021	0.00609	0.3451	0.0321	0	0.636	98	0.4
<i>Microstomus kitt</i>	0.00157	0.00318	0.4952	0	0.031	0.025	98.2	0.2
<i>Syngnathus acus</i>	0.00152	0.00305	0.497	0	0.0225	0.024	98.5	0.3
<i>Liparis liparis</i>	0.00134	0.00261	0.5115	0	0.02	0.465	98.7	0.2
<i>Triglidae</i>	0.00128	0.00366	0.3499	0	0.0275	0.058	98.9	0.2
<i>Trisopterus minutus</i>	0.00117	0.00337	0.3465	0	0.0143	0.049	99.1	0.2
<i>Hyperoplus immaculatus</i>	0.00112	0.00321	0.3482	0	0.0168	0.042	99.3	0.2
<i>Syngnathus rostellatus</i>	0.0011	0.00209	0.5253	0	0.0166	0.024	99.4	0.1
<i>Ammodytes tobianus</i>	0.00108	0.00315	0.3435	0.0146	0	0.877	99.6	0.2
<i>Pollachius pollachius</i>	0.00104	0.003	0.3478	0	0.0147	0.052	99.8	0.2
<i>Lophius piscatorius</i>	0.00093	0.00269	0.3462	0.0157	0	0.662	99.9	0.1
<i>Ammodytidae</i>	0.00057	0.00164	0.3478	0	0.0081	0.052	100	0.1

