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24 **Abstract**

25 This study investigated the cumulative impact of weirs on the downstream migration of wild
26 Atlantic salmon (*Salmo salar*) smolts in the River Foyle, Northern Ireland. In spring of 2013
27 fish were released in two tributaries of similar length; one tributary (impacted) had seven low
28 head weirs along the migration pathway and the other was devoid of such structures (un-
29 impacted). Salmon smolts fitted with acoustic transmitters were monitored via a passive
30 acoustic telemetry array during downstream migration. In 2014 the study was repeated only in
31 the impacted tributary. Overall freshwater survival rates were high (>94%). There was no
32 significant difference in mortality, movement pattern, delay or travel speeds between rivers or
33 between years at any phase of migration. Escapement of salmon smolts through Lough Foyle
34 (a marine sea lough) to the open ocean was low, approximately 18% in each year. Escapement
35 did not differ between impacted and un-impacted rivers. This study showed no post-passage
36 effects of weirs on mortality, migration speed or escapement of downstream migrating smolts.
37 This suggests that the elevated mortality at low head obstacles described in other studies is
38 not inevitable in all river systems. Migration through rivers with natural riffle-pool migration
39 may result in similar effects as those from low-head weirs. Causes of apparent high mortality
40 in the early part of marine migration in this study, are unknown; however similar studies have
41 highlighted the impact of fish predators on smolts.

42

43 Key Words: *Salmo salar*, Habitat fragmentation, River barriers, Downstream
44 migration, Survival

45

46

47 **Introduction**

48 Habitat corridors, which connect larger pieces of habitat together within a dissimilar matrix
49 are essential in facilitating gene pool coherence, recolonisation post disturbance and
50 population recruitment (Beier and Noss 1998; Elosegi et al., 2010). Species decline and
51 extinction is often preceded by the fragmentation of its distribution (Ceballos & Ehrlich 2002;
52 Baguette et al., 2013). Terrestrial connectivity enables animals to cross from one habitat patch
53 to another, often using one of several paths. In aquatic riverine habitats however, longitudinal
54 movement, along the river channel, tends to be dominant (Cote, Kehler, Bourne, & Wiersma,
55 2009) although in floodplain reaches, lateral movements are sometimes imperative (Lucas &
56 Baras 2001). Hydrological connectivity and the water-mediated transport of organisms,
57 energy and matter, is thus critical to ecosystem functioning. Species that migrate within river
58 habitats and between river and ocean habitats (e.g. anadromous and catadromous fishes) are
59 inevitably highly vulnerable to river corridor fragmentation.

60 In-river structures, both natural and artificial, such as waterfalls, dams, weirs, fords,
61 and culverts can have major impacts on fish communities, preventing free movement along
62 the riverine corridor (Baras et al., 1994; Lucas & Frear 1997; Jager et al., 2001; O'Hanley &
63 Tomberlin 2005; Kemp et al., 2008). It is estimated that in England and Wales alone there are
64 25,000 in-river, man-made, obstructions, of which 3,000 are significant and require mitigation
65 in order to meet objectives set by the Water Framework Directive (Directive 2000/60/EC),
66 and EU Eel legislation (EC No. 1100/2007) (Environment Agency 2009).

67 The impacts of large engineered in-river structures (>5 m head height; predominantly
68 hydropower dams), particularly on fish populations and assemblages is well documented
69 (Gowans et al., 2003; Antonio et al., 2007; Meixler et al., 2009; Branco et al., 2012). The
70 effects of low-head obstacles (<5 m head height) has however received much less attention,
71 yet they too have also been shown to have serious implications for fish passage (Lucas &

72 Frear 1997; Ovidio & Philippart 2002; O'Connor et al., 2006; Gauld et al., 2013).
73 Determining the likelihood of fish passage at river obstacles is highly complex because of the
74 numerous environmental and biological variables that may influence passage. The swimming
75 and leaping capabilities of fish of different sizes and species, as well as the heterogeneity of
76 environmental variables associated with riverine systems, such as flow and temperature, all
77 affect the probability of successful barrier (natural or man-made) passage (Baras & Lucas
78 2001). As such, any single barrier may prevent migration, cause a temporary delay in
79 migration, or have no effect whatsoever depending on the environmental conditions and
80 organism's biology. Passage at small scale barriers is likely to be highly temporal and
81 defined by changing environmental conditions, particularly flow (Kemp & O'Hanley 2010).
82 Such barriers are likely to be permeable to some species or some individuals of that species,
83 for example to a few size classes (Lucas & Frear 1997, O'Connor et al., 2006; Lucas et al.,
84 2009), resulting in temporary and variable delays to migration.

85 Downstream migration patterns of fish over small scale obstacles remains relatively
86 poorly described and quantified, however the reluctance of fish to progress downstream when
87 confronted with an in-stream barrier has been documented (Haro et al., 1997; Jepsen et al.,
88 1998). Elevated mortality resulting from physical damage during passage through
89 hydropower turbines is regularly reported (Hvidsten & Johnsen 1997; Thorstad et al., 2012a).
90 It is also possible that physical damage of fish occurs from downstream passage of over-spill
91 weirs, through contact with the weir face or stream bed due to hydraulic forces present at such
92 structures. This impact, not necessarily causing instant mortality, may result in a delayed
93 response, affecting individuals during the later migration. Thus to fully understand the impact
94 of low head impoundments and how these man-made structures compare with passage within
95 a natural system without engineered structures, it is essential to understand post-passage
96 impacts in addition to pre-passage behaviour (Roscoe et al., 2011).

97 Migration delays and increased mortality have been shown in downstream migrating
98 anadromous trout (*Salmo trutta*) smolts over a single low head weir of 3m in height (Gauld et
99 al., 2013). This study showed mortality rates of between 9% and 44% of tagged fish
100 associated with a single weir and that the mortality rate was highly dependent upon flow rate.
101 Even mortality rates from the lower end of the range recorded by Gauld et al., (2013), point
102 towards a potentially high cumulative loss over several low-head obstacles in series. The
103 measurement of this cumulative impact for small engineered structures is rare, although it has
104 been demonstrated for medium-sized and larger obstacles (Gowans et al., 2003; Holbrook et
105 al. 2011). However the idea that delayed migration in general can have serious negative
106 impacts is commonly expressed (Chanseau & Larinier 1999; Naughton et al., 2005; Caudill et
107 al., 2007; Holbrook et al., 2011). Downstream migrating smolts are subjected to predation
108 from mammalian, avian and fish predators, where the impact of a barrier is a delay or an
109 overall reduction in travel speed during migration, this can negatively impact upon survival
110 through increased exposure to predation risks (Jepsen et al. 1998; Koed et al. 2002). A
111 number of studies on salmonids indicate a positive correlation between migration success and
112 migration speeds through entire systems (Chanseau & Larinier 1999; Naughton et al., 2005,
113 Holbrook et al., 2011).

114 There is a paucity of studies that have examined smolt migration in pristine or natural
115 systems (Welch et al., 2008), thus information on natural migration speeds, delay and
116 particularly mortality resulting from natural riverine structures, such as rapids, pools and
117 riffles, is lacking. Studies on impacted rivers alone also lack any credible control against
118 which to test migration behaviour; such information would allow any direct effect of riverine
119 barriers to be assessed in terms of delayed migration or mortality within regulated rivers
120 (Thorstad et al., 2007).

121 Only recently has technology become available that allows us to address some of these
122 behavioural questions. Telemetry enables the real-time movement of fish to be studied,
123 allowing the environmental factors which enable migration or cause delay to be measured,
124 whilst at the same time assessing mortality and migration success. The study presented here,
125 used acoustic telemetry and a comparative approach to compare seaward migration of
126 Atlantic salmon smolts in adjacent tributaries: one with no man-made obstacles; the second
127 with seven, low head, man-made obstacles in series.

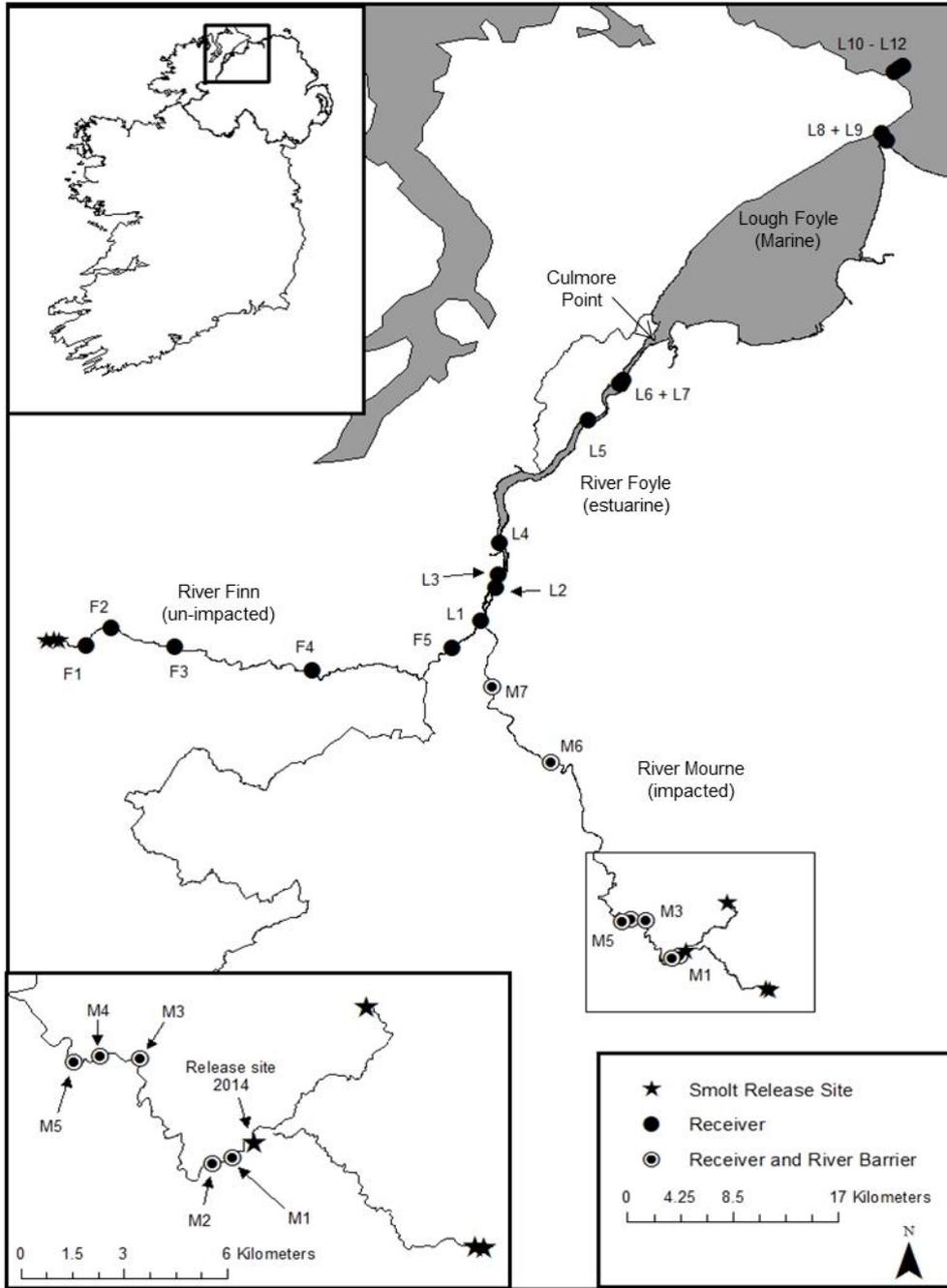
128 It was hypothesised that the cumulative effect of low-head, but passable, barriers would
129 be to reduce travel speed, increase mortality rate and lower escapement success of seaward
130 migrating Atlantic salmon smolts, by comparison to those in a neighbouring river without
131 such obstacles.

132

133 **Methods**

134 *Study Area*

135 The study was carried out in the River Foyle system (55°00'N; 07°20'W). The river has a
136 catchment area of 4450 km² and forms part of the border between the Republic of Ireland and
137 Northern Ireland (UK) (Fig. 1). The whole Foyle system is designated an EU Special Area of
138 Conservation (SAC) for Atlantic salmon. There are two main tributaries within the catchment;
139 the River Finn, which is free from anthropogenic river obstacles apart from a single fish
140 counting weir (between F4 and F5), the form of which has been shown to have no impact on
141 upstream fish movement (Smith, Johnstone, & Smith, 1997). In contrast, the second major
142 tributary, the River Mourne, has seven man-made low-head overspill weirs along its length
143 (Fig. 1, Table 1). All barriers span the complete river width and had water flowing over them
144 continuously during the study period (albeit the depth varied with time). Here the Rivers Finn



145

146 Figure 1: Location of the Foyle catchment in Ireland, on the border between Northern Ireland
 147 and the Republic of Ireland (top left). Automatic listening station (ALS) deployment
 148 throughout the catchment is presented in the main map. Bottom left is a larger version of the
 149 headwater of the impacted river where river barriers and release sites are in close proximity.
 150 River flow is in a northerly direction, the River Foyle is tidal downstream from the confluence
 151 of Rivers Finn and Mourne (L1).

152 and Mourne will be referred to as ‘un-impacted’ and ‘impacted’ rivers, respectively. The
153 confluence of these two rivers form the upper reach of the tidal River Foyle and represents a
154 transitional/estuarine habitat with surface salinity levels (Practical Salinity Units [PSU]) at its
155 most upstream point (L1, Fig.1) averaging 0.14psu, increasing to 26.6psu at Culmore Point,
156 where the river enters a large sea lough, Lough Foyle (Fig. 1). The section from the
157 confluence of the un-impacted and impacted tributaries to the entry of the sea lough, will be
158 referred to as ‘estuarine.’ Lough Foyle salinity levels average 26psu at its most inland
159 location (Culmore Point - where it is strongly influenced by freshwater run-off) to 35psu at its
160 most northerly point where salinity rarely falls below 32psu (salinity data provided by
161 Department of Environment Marine Environment Division, Northern Ireland). The Lough
162 Foyle section will be referred to as a ‘sea lough’ and classified as the early marine phase
163 migration for emigrating salmon smolts.

164 ***Smolt capture and tagging***

165 This study was conducted across two years. In 2013 fish were tagged in both the impacted
166 and un-impacted rivers. Unexpectedly (*cf* literature, see above), in 2013, freshwater survival
167 was high in the impacted river and there was no significant difference in travel speeds in
168 freshwater between the impacted and un-impacted rivers. Therefore, in 2014, to determine if
169 the same pattern held, the study was repeated in the impacted river. Due to resource
170 limitations, tagged fish were released only in the impacted river.

171 In 2013, salmon smolts were captured by electro-fishing in the upper reaches of both
172 rivers between the 14th and 15th April. Due to technical problems, salmon smolts were
173 captured by rod and line in April 2014. Smolts were placed into a holding tank filled with
174 aerated river water. Fish large enough for tagging (>15g) and which were also clearly
175 smolting, were anaesthetised with clove oil (0.5mg per litre); mass (g) and fork length (FL,

176 mm) were recorded prior to being placed on a v-shaped surgical pillow saturated with river
177 water. An incision (11-13mm) was made along the ventral abdominal wall anterior to the
178 pelvic girdle. A coded acoustic transmitter (either, Model LP-7.3, 7.3mm diameter, 18mm
179 length, 1.9g weight in air, Thelma Biotel AS, Trondheim, Norway [2013], or Model V7-2x, 7
180 mm diameter, 18 mm length, 1.4 g weight in air, Vemco Ltd, Nova Scotia, Canada [2014])
181 was inserted into the peritoneal cavity. The incision was closed with two independent sterile
182 sutures (6-0 ETHILON, Ethicon Ltd, Livingston, UK). Fish were aspirated with 100% river
183 water throughout the procedure. Tags were programmed to have an acoustic transmission
184 repeat cycle of $30s \pm 50\%$, giving a tag life span in excess of 90 days.

185 On completion of tagging, fish were placed into a recovery bucket filled with aerated
186 river water and allowed to recover before being placed into a keep box which was positioned
187 in-river overnight. No mortality occurred at any stage throughout the tagging period. Fish
188 were released the day after tagging close to their capture site within their respective tagging
189 groups (Fig. 1).

190 *Acoustic Tracking*

191 Movement of tagged smolts was determined using fixed position automatic listening stations
192 (ALS) (Vemco: VR2W). All ALS were deployed prior to tagging and release of fish, ALS
193 were recovered in July of each year, after the migration period and the expected tag life had
194 been reached. Six ALS were positioned in the impacted river (M1 – M7), each located
195 slightly upstream from a river obstacle (Fig. 1). All such structures were over-spill sloping
196 weirs, apart from M1 which comprised a degraded historic weir and a series of rapids and M6,
197 a vertical weir. Barriers ranged from 0.75-4.3m head height (Table 1).

198 Five ALS were assigned to the un-impacted river (F1 - F5), located at deep holding
199 pools or glides where river flow was generally slow and similar to the conditions created

200 artificially above man made obstacles (i.e. deep, slow moving impounded water located
201 immediately upstream of riverine barriers) (Fig. 1). An additional four ALS were positioned
202 downstream of the confluence of the study rivers (L1 – L4) at the tidal limit of the River
203 Foyle. To ensure adequate spatial coverage and detection of emigrating smolts from both
204 rivers, data from these were combined to create a single detection zone henceforth named L4.
205 A further three ALS were located downstream within the estuarine part of the River Foyle (L5
206 - L7). Entrance to the sea lough was defined as detection at L6 or L7. Two final receivers
207 covered the exit from the Sea Lough into the Atlantic Ocean with successful early marine
208 migration being defined as detection at either L8 or L9.

209 Range tests were undertaken throughout the array to ensure complete receiver
210 coverage at each location, providing a detection gate through which tagged individuals had to
211 pass. More specifically at ALS L8 and L9 (Fig.1), to ensure detection coverage was adequate
212 to detect passing tags, an acoustic tag (Model LP-7.3, 139dB re 1 μ Pa power, Thelma Biotel
213 AS, Trondheim, Norway 2013) was suspended at 3 m depth and trolled for 1500m by a
214 drifting boat (engine off) to test for acoustic breaches, this was repeated four times. Data from
215 this exercise identified an effective acoustic range of 450m and thus receivers were deployed
216 to create overlap in the detection ranges of ALS L8 and L9. Tag failure rate reported by
217 manufacturers (Vemco, Thelma) is low (<2%). For Thelma tags of the same model used here
218 Gauld et al. (2013) reported control tag failure rates of 0% in field tests. In 2014, three
219 receivers were also located in a transect stretching 2 km out from the North coast of Ireland,
220 adjacent to Lough Foyle (L10 – L12, Fig. 1).

221 Here, freshwater migration is defined as the movement of tagged fish from the most
222 upstream receiver (M1 or F1) downstream to L4. In 2014, receivers L1 to L4 were removed
223 for logistical reasons, and freshwater migration in the impacted river was calculated as
224 occurring between M1 and M7 in 2014. It is assumed that fish which were detected at the first

225 upstream receivers (M1 or F1) but not detected leaving freshwater, died within the freshwater
226 section and are thus defined as freshwater mortalities. This is a reasonable assumption as de-
227 smoltification is rare in Atlantic salmon smolts (McCormick, Hansen, Quinn, & Saunders,
228 1998). Successful estuarine migration is defined here as the movement of fish between L4 and
229 L6 + L7 in 2013 and between M7 and L6 + L7 in 2014 (due to the removal of L4), similarly
230 fish that were detected at L4 (M7 in 2014) but not at L6 + L7 are assumed to have died within
231 the estuary (estuarine mortality). Successful early marine phase migration is defined as
232 movement between L6 or L7 to where the lough discharges into open sea (L8/L9), finally fish
233 detected at L6 + L7 but not at L8/L9 were assumed to have died within the sea lough (early
234 marine mortality).

235 Freshwater travel time of smolts was calculated as the time between the last detection
236 at receiver M1 or F1, and first detection at the estuarine receiver L4 (M7 in 2014). Estuarine
237 travel time was calculated as the time from the last detection on L4 (M7 in 2014) until the
238 first detection at L6 or L7. Data from 2013 for the impacted river were recalculated to
239 account for receiver location change (removal of L4 in 2014) i.e. freshwater travel calculated
240 as M1 to M7 and estuarine travel as M7 to L6 or L7 (same distances at 2014), enabling a
241 direct comparison between years. Analysis was thus conducted both spatially, within one year
242 (impacted vs un-impacted, 2013) and temporally (impacted 2013 vs impacted 2014).

243 Distance travelled between detection sites was calculated using the centre line of the river
244 with ARC GIS software. It is recognised that this is not the shortest or longest possible route
245 an individual may use; however it is likely to be representative of the actual migration
246 distance. Freshwater travel distance in the impacted river (M1 – L1) was 50 km, 16% longer
247 than the un-impacted river (F1 – L1) survival results are reported on a kilometre by kilometre
248 basis and migration speed in km.d^{-1} to reflect this variation.

249 ***Environmental data***

250 River flow data for the rivers were provided in the form of discharge data for the impacted
251 river (provided by the Department of Agriculture and Rural Development, Northern Ireland),
252 and stage (used as a proxy for discharge, provided by the Office of Public Works, Ireland) for
253 the un-impacted river. Mean daily discharge from the impacted river was used to assess flow
254 conditions for the study period in both 2013 and 2014. Data from the previous ten years were
255 also analysed to identify long term trends in river flow for the impacted river (Fig. 3).

256

257 ***Statistical Analysis***

258 All analysis was performed using R statistical software programming. Welch-t-tests were
259 used to test for differences in fork length between populations. Normality of data was
260 confirmed using a Shapiro Wilks test. Where normality was not confirmed or assumptions of
261 t-tests not met, Wilcoxon Mann-Whitney rank sum tests were performed. Wilcoxon Mann-
262 Whitney rank sum tests were also performed on differences in delay times between rivers and
263 speed of travel due to some observations highly skewing the mean observation. Fisher's exact
264 tests were used to determine if the observed frequencies of mortalities was different from
265 expected frequencies between years, rivers and phases of migration. Analysis of variance
266 (ANOVA) was used to determine differences in delay by fish between each of the barriers,
267 data were log transformed to meet assumptions of normality, confirmed by Shapiro Wilks
268 test. A Levene's test was used to determine the differences in variances of freshwater
269 migration speed between impacted and un-impacted rivers.

270 **Results**

271 Sixty eight fish were tagged during the study period: impacted 2013, $n = 20$, (mean fork
272 length [FL] = $144.3 \pm \text{SD } 9.1$, mean mass [M] = $31.3 \pm \text{SD } 4.9\text{g}$) un-impacted 2013, $n = 19$,

273 (mean FL = $132.2 \pm \text{SD } 10.8$, mean M = $24.8 \pm \text{SD } 6.3\text{g}$), impacted 2014, $n = 29$, (mean FL
274 = $135.2 \pm \text{SD } 27.3$, mean M = $28.8 \pm \text{SD } 7.0\text{g}$). There was a significant difference in fish
275 length between rivers (*t test*, $t = 2.94$, $p = 0.005$, d.f. = 36.5,) but no difference in length
276 between years ($t = 1.49$, $p = 0.14$, d.f. = 46.9) (Table 1). Data from the ALS receiver array was
277 used to estimate survival for all fish over multiple sections along their migration. Data from
278 ALS M5 were removed from the analysis because acoustic noise severely reduced detection
279 efficiency throughout the study period. Fish which were not detected at the first receiver
280 within the array (M1, F1) were eliminated from all further analysis. A lower proportion of
281 fish (41%, $n = 12$) were detected within the array in 2014 compared to 2013 (85%, $n = 17$) in
282 2013. There was no difference in fork length or tag mass to body mass ratios between fish
283 detected within the array and those not detected. The exact fate of undetected fish cannot be
284 directly determined. No smolt was detected at a downstream receiver which was not
285 previously detected at an upstream receiver

286 Total escapement (survivorship of fish from first upstream detection zone [M1, F1] to
287 the lough exit to the open coast at either L8/L9) of tagged fish in 2013 was 18% ($n = 3$), and
288 19% ($n = 3$) from the impacted and un-impacted river respectively (Fig. 2). In 2014, loss of
289 ALS L8 prevented total coverage of the lough exit and thus full escapement cannot be
290 determined. A single fish was detected at L9, with no individuals detected at L10 - L12 thus
291 at least one individual did reach the open ocean. Data from 2013 indicates that 50% of fish
292 were detected at either receiver (detection probability of 50%) at L8 and L9. Thus a cautious
293 estimation may indicate two fish likely successfully migrated to the open ocean in 2014.

294 Freshwater survival within the un-impacted river (100% per km, $n = 17$) was not
295 statistically different ($p = 0.53$, *Fisher's Exact Test*) from the impacted system (99.9% per km)
296 in 2013. No difference in the number of mortalities between years ($p = 0.62$, *Fisher's exact*
297 *test*) was observed for the impacted river. Survival rates were marginally lower during

298 estuarine migration for tagged fish from both rivers (impacted 2013 = 99.4% per km, un-
299 impacted 2013 = 99% per km) in 2013 (Fig. 2). Significantly lower survival ($p < 0.01$,
300 *Fisher's Exact Test*) occurred in the early marine phase of migration (L6 + L7 to L9) in both
301 rivers (impacted 2013 = 97.4% per km, un-impacted 2013 = 97.5% per km) and years
302 (impacted 2014 = 97.3% per km), than in the freshwater and estuarine phase (L1/F1 to L6 +
303 L7 [Fig. 2]).

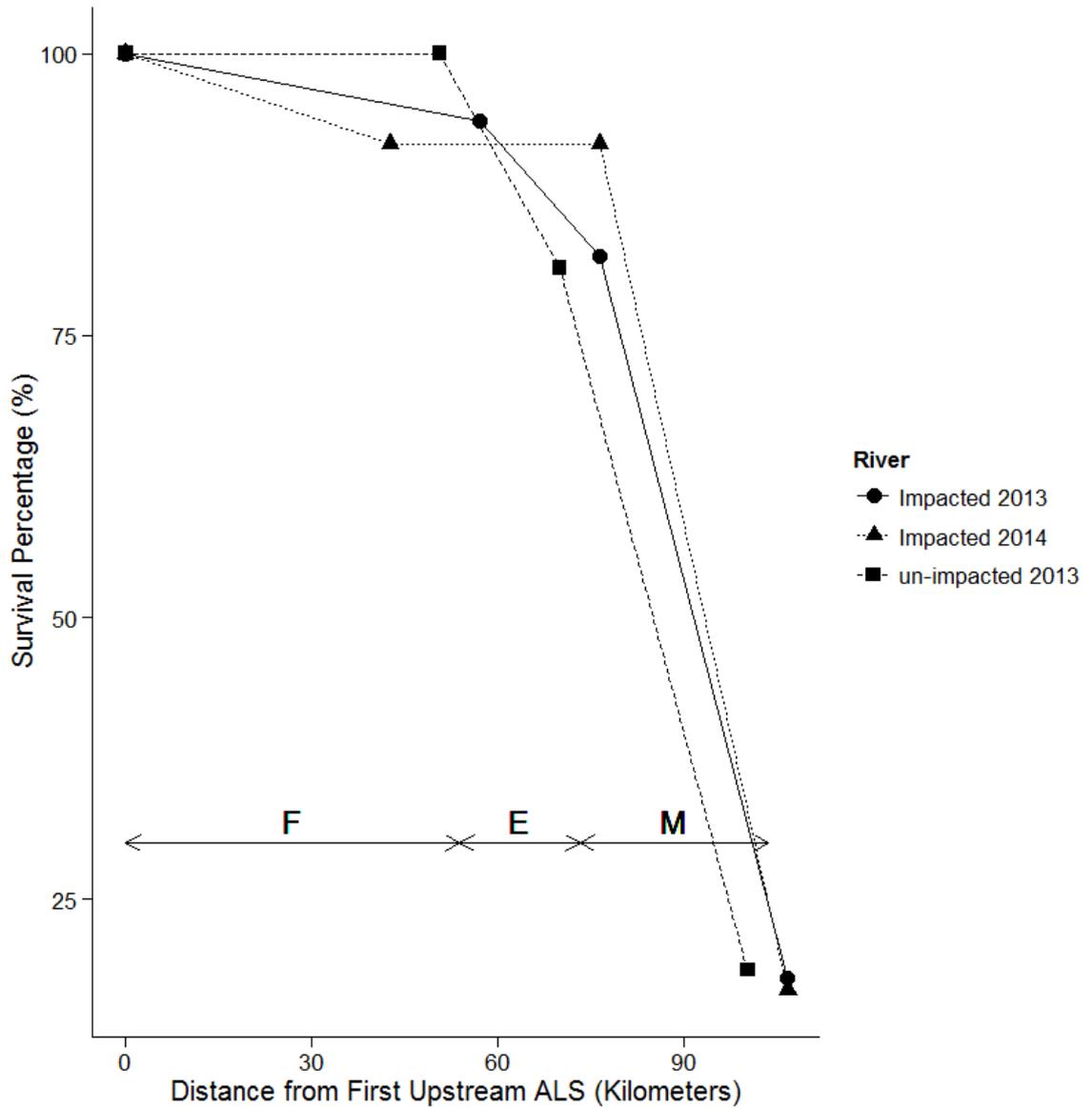
304 ***Migration Delay***

305 Delay, a measure of how long an individual fish remained in the upstream vicinity of a
306 potential manmade (impacted) or within a natural (un-impacted) pool was calculated as the
307 time between first and last detection at each individual freshwater ALS, located immediately
308 upstream of a weir (impacted river) or within a natural pool (un-impacted river) for each
309 individual. Mean delay per fish in 2013 was not significantly different between the un-
310 impacted river ($n = 18$, median = 0.16hr, range 0-18.2hr) and impacted river ($n = 17$, median
311 = 0.17hr, range 0-126.74hr) (Wilcoxon-Mann-Whitney, $W = 159$, $p = 0.86$). Mean delay in
312 2014 in the impacted river ($n = 12$, median = 0.5hr, range = 0-72.5hr) was not significantly
313 different than in 2013 ($W = 84$, $p = 0.44$). Total Delay (sum of delays at individual receivers,
314 per fish) at some individual obstacles (Table 1) within the impacted river was significantly
315 different between years (M3, $W = 29$, $p = 0.03$; M4, $W = 24$, $p = 0.03$, M7, $W = 85.5$, $p =$
316 0.03) but not at others (M1, M2, M6).

317 Analysis of variance (ANOVA) testing identified no difference in delay between
318 individual obstacles for the un-impacted river ($F [4,15] = 1.4$, $p = 0.3$) or impacted river in
319 either 2013 ($F [5,57] = 1.8$, $p = 0.1$) or 2014 ($F [5,62] = 0.7$, $p = 0.6$). Two individuals in
320 2013 were delayed for 118 and 126 hours respectively at M2, creating outliers that
321 exaggerated the mean delay time from that measured for other fish (Table 1. Median delay at

322 M2 = 0.07hrs). Similarly two fish in 2014 were delayed for 49 and 72 hours compared to a
323 median of 0.16hrs (Table 1).

324



325

326 Figure 2: Survivorship curve of tagged salmon smolts from the three release groups.
327 Survivorship is calculated for freshwater (F), estuarine (E), and early marine (M) elements of
328 the migration. Distance 0 is the most upstream ALS with distances calculated downstream
329 from this point.

330

331

332

333 Table 1: Summary of obstacle type with mean and median time of fish detected at ALS
 334 deployments across the study period. Time is not calculated at M5 due to receiver being
 335 compromised by excess noise.
 336

Station name	Obstacle type	Head height (meters)	Mean (Median) delay (Hours)	
			2013	2014
F1	N/A	N/A	0.06 (0.02)	NA
F2	N/A	N/A	0.17 (0)	NA
F3	N/A	N/A	0.18 (0.008)	NA
F4	N/A	N/A	0.08 (0.08)	NA
F5	N/A	N/A	1.97 (0.38)	NA
M1	Broken weir above rapids	4.3	1.18 (0.05)	6.17 (0.06)
M2	Sloping Weir	0.75	18.86 (0.07)	5.48 (0.16)
M3	Sloping Weir	1.89	0.18 (0.14)	0.56 (0.31)
M4	Two sloping weirs approx. 30 meters apart	1.5+ 0.75	0.15 (0.11)	6.21 (0.97)
M5	Over spill weir	0.75	NA	NA
M6	Vertical weir	1.2	0.07 (0.07)	0.04 (0)
M7	Sloping weir	3.4	0.86 (0.22)	0.06 (0.03)

337

338 *Freshwater Migration*

339 Ground speed was highly variable within river groups. The range in ground speed for the un-
 340 impacted river was 2.3 – 17.3 km.d⁻¹ and for the impacted river 1.8 – 103.3 km.d⁻¹ across both
 341 years.

342 Freshwater ground speed in 2013 in the impacted river (mean ± SD, 17.2 ± 22.6,
 343 median = 10.6 km.d⁻¹) was not significantly different (Wilcoxon rank sum, $W = 145$, $p = 0.34$)
 344 to that of the un-impacted river (mean ± SD 6.4 ± 4.4, median = 4.6 km.d⁻¹). One fish
 345 travelling at 41.8 km.d⁻¹ skewed the mean in the impacted river but was included within the
 346 Wilcoxon test. Freshwater ground speed in 2014 was not significantly different to 2013
 347 (Wilcox rank sum, $W = 179.5$, $p = 0.37$). A Levene's test indicated no significant difference in

348 variances of ground speed between impacted and un-impacted rivers ($F = 3.46, p = 0.07$) or
349 between years in the impacted rivers ($F = 0.53, p = 0.47$).

350 *Estuary and Early Marine Migration*

351 Mean travel time of fish migrating through the estuary was 75 hrs (range 11 hrs – 20 days) at
352 a mean speed of 15 km.d^{-1} (range = $0.9 - 52 \text{ km.d}^{-1}$). There was no significant difference in
353 estuarine ground speed between rivers ($W = 105, p = 0.06$) or between years ($W = 114, p =$
354 0.54). There was no significant difference between freshwater or estuarine ground speeds ($t =$
355 $0.013, p = 0.99$).

356 Data on movements within the sea lough are limited to six individuals in 2013. Mean
357 travel time through the sea lough (30 km) was 59 hrs with a mean ground speed of 19.4 km.d^{-1}
358 (range = $4.9 - 48.1 \text{ km.d}^{-1}$). A single individual was successful in reaching L9 in 2014 and did
359 so in 30 hrs at a speed of 24 km.d^{-1} .

360 *Inter-annual variation in River Discharge*

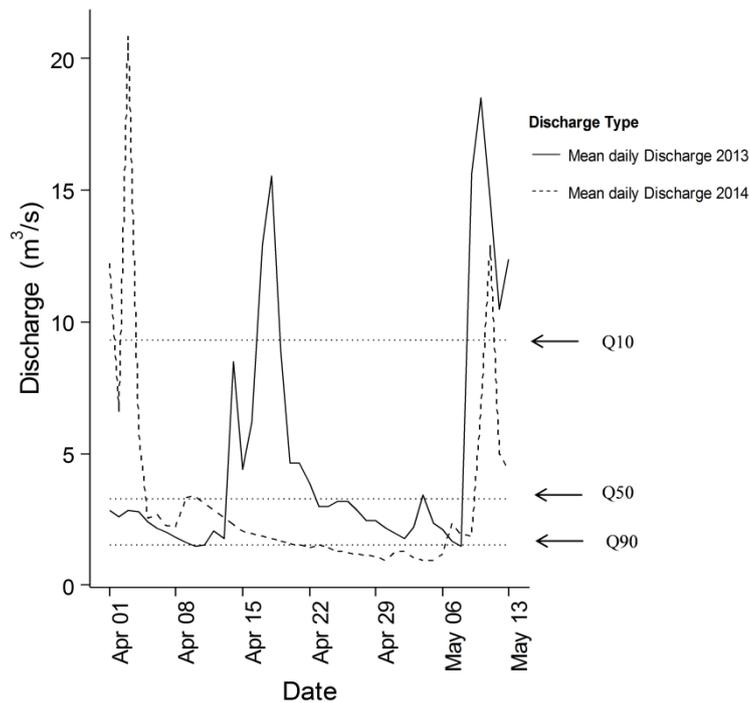
361 River discharge between the two study years contrasted markedly. Flow in the Mourne
362 (impacted river) in 2014 fell below the Q90 exceedance for an extended proportion (16 days)
363 of the migration period, compared to 2013 when it fell below this level only for three days.
364 Indeed river flow in 2013 was considerably higher with seven days being above Q90
365 compared to only three in 2014. A peak in discharge in mid-April, 2013 sustained moderate
366 flows throughout the migration period. No such peak was present in 2014 resulting in
367 declining low flows from 10th April through to May 6th (Fig. 3).

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372
 373 Figure 3: Mean daily flow taken from flow gauging station on the impacted river for 2013 and
 374 2014. Also drawn are flow exceedance percentiles, Q90, Q50 and Q10 flows calculated from
 375 mean daily flows of the previous ten years of data during the study period.
 376

377 **Discussion**

378 This study is the first to compare directly downstream wild Atlantic salmon smolt migration
 379 in a river impacted by multiple low head obstacles, with a river un-impacted by such
 380 structures in a single catchment and thus subject to the same general environmental
 381 conditions. Surprisingly, survival rates during the freshwater phase of migration in the
 382 impacted river were high across both years (93%). There was no evidence of differential
 383 survival rates between impacted and un-impacted rivers in the one year where this
 384 comparison was possible (2013). Whilst acknowledging the modest sample size, this finding
 385 contrasts significantly with a number of other studies that indicate that in-stream obstructions,

386 including low head ones, contribute to smolt mortality and ultimately reduce smolt
387 escapement (Aarestrup & Koed 2003; Thorstad et al., 2012a; Gauld et al., 2013). Similarly, it
388 has been shown recently that survival rates for Pacific salmon (*Oncorhynchus* species) smolts
389 is higher in rivers which lack large hydro-electric dams (Welch et al., 2008). There are a
390 number of environmental conditions that have the potential to impact upon, migrating salmon
391 and it is highly likely that these differ between catchments. Similarly it is highly likely that
392 barrier effects on smolts might reasonably be expected to be site and catchment specific.

393 The freshwater survival rate of Atlantic salmon smolts for the impacted river in this
394 study is broadly in line with that reported in UK rivers with no anthropogenic barrier effects.
395 For example a study in the River Conway, UK, reported survival of 99.4% km⁻¹ (Moore et al.,
396 1995); in the River Test, UK, 95% km⁻¹ was reported (Moore et al., 1998) and in a meta-study
397 (Thorstad et al., 2012) found survivorship in the range 93% - 99.7% km⁻¹. The barriers in this
398 study appear similar in format (1-3m head height, overspill weirs) to those described by
399 Gauld et al., (2013) yet mortality rates between the two studies contrasts considerably. It is
400 likely local pressures, such as predation, influence survival differentially across catchments.
401 Salmon populations exhibit both ecological and genetic differences between rivers; it is
402 possible that populations might exhibit local adaptations to their the natal water body (Taylor
403 1991; Heinimaa et al., 1998; Garcia de Leaniz et al., 2007). In this study there were no
404 differences in mortality between smolts migrating from contrasting rivers during the estuarine
405 migration phase. Thus at least in this study there is no evidence of delayed post-passage
406 effects of low head impoundments on downstream migrating smolts.

407 Despite high freshwater and estuarine survival, overall escapement to sea (18%) was
408 relatively low when compared with other studies of river and estuarine smolt migration. For
409 example in the River Tweed, UK between 19 and 45% was recorded (Gauld et al., 2013); in
410 Nova Scotia, Canada, similar escapement was 39-74% in one study (Halfyard et al., 2012); in

411 the River Lærdalselva, Norway, this was 85% (Urke et al., 2013) and in the Romsdalsfjord
412 System, Norway 35% , (Thorstad et al., 2007). Lough Foyle contains a number of marine fish
413 species, of which spurdog (*Squalus acanthias*) are thought to be present in high densities.
414 Spurdog are a known predator of Pacific salmon smolts (*Oncorhynchus* species) in the Strait
415 of Georgia, and are also a significant source of mortality for seaward migrating smolts; a
416 single individual having been recorded with 17 smolts within its gut (Beamish et al., 1992;
417 Friedland et al., 2012). Previous studies in Norway estimated that cod (*Gadus morhua*) were
418 taking 24.8% of Atlantic salmon smolts from the River Surna (Hvidsten & Møkkelgjerd
419 1987). Similarly cod and saithe (*Gadus virens*) populations combined were responsible for
420 20% of smolt mortality in the River Orkla (Hvidsten & Lund 1988). These and other gadoid
421 species are present within Lough Foyle (McGonigle et al., 2011), yet there is little
422 information available on other predator species, such as birds or mammals, or on population
423 numbers of potential predators and their diet. Thus it is difficult to directly quantify the effect
424 of predators on smolt emigration, particularly in areas such as sea loughs and river mouths
425 where predator density is likely to be high and sea migrating smolts may be constrained by
426 geography (Larsson 1985; Greenstreet et al., 1993; Dieperink et al., 2002; Woody et al., 2002;
427 Serrano et al., 2009; Thorstad et al., 2012).

428 The fact that survival was not affected by annual variations in flow is somewhat
429 surprising. Exceedingly low flows experienced by migrating smolts in 2014 (18 consecutive
430 days below Q90) apparently did not impact on mortality, migration speeds or delay in
431 freshwater migration when compared with data from a hydrologically typical year in 2013. In
432 contrast, an extended low flow period of 18 days below Q95 in the river Tweed resulted in
433 44% of smolts failing to pass a single barrier, compared to 9% failure in a ‘normal’ spring
434 (Gauld et al., 2013). Despite studies identifying a positive relationship between flow and
435 smolt survivorship at both large barriers (Kjelson & Brandes 1989; McCormick et al., 1998)

436 and small scale barriers (Gauld et al., 2013), results of the study presented here contrast
437 markedly with these earlier findings. Slack waters above weirs and dams likely create suitable
438 habitat for predatory behaviour that does not normally occur in fast flowing river stretches.
439 Any delay caused by barriers potentially expose fish to predators for a greater period of time
440 thus increasing exposure to potential predators. Although telemetry tagging effects on fish
441 behaviour can occur (Wilson et al., 2017), in this study if any such effect occurred, it was
442 likely to be expressed equally between impacted and un-impacted rivers as the same method
443 was used. The main findings of this study, that survival was high and not different across
444 sites, suggests no obvious tagging effect. Tag effects from the same study system have been
445 explored in a previous paper (Newton et al., 2016). Taken together and in the context to
446 relevant contemporary literature (Cooke et al., 2011; Jepsen et al., 2008; Larsen et al., 2013;
447 Wagner et al., 2011; Brown et al., 1999; Rechiskey and Welch 2010) we conclude that there
448 was no obvious tagging effect resulting in bias in our study.

449 Delay and mortality at riverine barriers is regularly reported, however the direct
450 simultaneous comparison of delay in an impacted river to that of a natural system is rare
451 (Thorstad et al. 2012a; Cooke & Hinch 2013). This study demonstrated that delays (or natural
452 'holding' behaviour) resulting from natural pools and impoundments to migration in natural
453 systems can be equivalent. Given that the findings presented here run contrary to several other
454 studies, we tested the magnitude of the effect for its proximity to statistical significance. Thus
455 we simulated a sequential increasing differential in the median travel speed between fish from
456 the two groups (in the impacted and un-impacted rivers) to identify the point where the
457 differential is large enough in magnitude to exhibit a statistically significant difference for $P=$
458 0.05. The result shows that the differential in modified travel speed would need to increase
459 from 0.07 ms^{-1} , almost two fold to 0.12 ms^{-1} to become statistically significantly different.
460 This points to the finding presented here and the conclusions drawn from this as being robust.

461 Site specific delays can differ significantly between years even when delay throughout the
462 whole system does not. Surprisingly, delay was not different between individual barriers
463 within years despite significant physical differences in barrier construction (Table 1). Because
464 of the existence of natural, but unpredictable, holding behaviour in un-impacted and impacted
465 river systems, it may not be feasible to directly compare downstream passage time of smolts
466 in an impacted reach to that of an un-impacted reach within the same river. Indeed what is
467 perceived as a delay above an obstacle may actually be a natural ‘holding’ pattern in a pool
468 created by the obstacle. Holding is a natural phenomenon and delay should be measured
469 across a whole emigration period and stream reach rather than at individual sites. Thus care
470 must be taken when attributing the cause of a delay solely to a man-made river obstacle.

471 A common limitation in telemetry studies, and applicable here, is that of low sample size,
472 the primary driver of which is transmitter cost. Individuals within a species may differ greatly
473 in their behaviour and behavioural response to environmental variables (Dall et al., 2012).
474 Thus it is sometimes difficult to determine whether results from small sample sizes accurately
475 reflect the wider population they represent. Low sample sizes must be contrasted with the
476 benefit of data collected which cannot be generated through other techniques. Although
477 sample size in this study is relatively small, the high survival rate of fish through freshwater
478 and estuarine portions, across years, supports the primary conclusions. Similarly despite the
479 low number of fish detected reaching the open ocean, mortality rate per kilometre is not
480 dissimilar to that reported in other studies of estuarine and marine migration (Thorstad et al.,
481 2007a). However there is an ever present need for similar telemetry studies with larger sample
482 size and longer time series. In reality, to accurately represent a significant proportion of any
483 smolt population may require thousands of individuals to be tagged due to the vast numbers
484 of downstream migrating juveniles. Although sampling strategies differed between years, the
485 low mortality observed in year 1 (2013) differs substantially from that reported elsewhere and

486 requires some interpretation (Lucas & Frear 1997; Ovidio & Philippart 2002; O'Connor et al.,
487 2006; Gauld et al., 2013). Variation in river flow between years has previously been reported
488 to affect smolt survival (Gauld et al., 2013). Repeating this study in the impacted river, across
489 years, enabled the effect of river flow to be eliminated as the cause of high survival. Resource
490 constraints however did not allow for a complete repetition (by virtue of a lack of a full
491 control group in the un-impacted river) of the previous year (2013), yet the similarities
492 between the data (high survival) suggest that survival within the system was generally high
493 and riverine barriers did not elevate mortality.

494 Our study raises important questions regarding the migration of Atlantic salmon smolts, in
495 that not all systems with multiple obstacles, although expected to have cumulative effects,
496 may in fact result in elevated mortality. The evidence of this study is that migration through
497 rivers with natural riffle-pool sequences may be no different to that of a system with low head
498 anthropogenic obstacles. It is clear there is a requirement for further studies, with greater
499 sample sizes, of natural migration of wild smolts in un-impacted rivers, before it is possible to
500 attribute mortality and delay to a direct consequence of weirs, dams and engineered in-river
501 structures.

502

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504

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510

511

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