Assessing the effects of forest-to-bog restoration in the hyporheic zone at known Atlantic salmon (*Salmo salar*) spawning sites

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SUMMARY

In the UK, large areas of blanket bogs were afforested with non-native conifers between the 1960s and the 1980s. Following recognition of the detrimental effects of such practice on biodiversity and carbon stocks, large-scale restoration trials started in the late 1990s and are further supported by recent changes in policy. The removal of forestry from peatlands is likely to be a widespread land-use change in the coming decades and could affect adjacent freshwater systems. This study aimed to investigate whether forestry removal with drain blocking affected nearby spawning sites used by Atlantic salmon (*Salmo salar*). We analysed the chemistry of hyporheic (beneath and just above the streambed) and surface water, and measured sediment deposition upstream of, within and downstream of a forestry block in the north of Scotland, during and after restoration management operations. We found no immediate effect of management except on potassium and zinc concentrations, which increased after restoration. The general lack of effect is attributed to catchment properties, including the small proportion of catchment (< 5 %) affected by management, and to dilution effects related to heavy precipitation during the intervention phase. We suggest that longer-term monitoring should be implemented as the sizes of areas undergoing restoration management increases.

KEY WORDS: dissolved oxygen, peatland, potassium, river, sediments

INTRODUCTION

In Scotland, recreational fisheries for Atlantic salmon (*Salmo salar*) are estimated to bring 50–100 million pounds sterling (~ 58–115 million euro as this article goes to press) per annum to the economy (McLay & Gordon-Rogers 1997). This represents a critical income for rural areas such as Highlands Region (Scottish Government 2014). Under the Conservation Regulations 1994, Atlantic salmon is also a protected species in freshwater, and often considered a ‘flagship’ species. Adult salmon lay their eggs under several centimetres of stream gravel in autumn and the eggs incubate in situ over winter. After the young fish emerge, they exploit open stream habitat in the same vicinity for two or three years before they become smolts and migrate to sea. Thirteen of the rivers where Atlantic salmon come to breed have their sources in the large expanse of blanket bogs that make up the Flow Country peatlands in the counties of Caithness and Sutherland.

Whilst it remains one of the least damaged peatland areas in the UK, the Flow Country still bears the imprint of centuries of human activities. One of the largest disturbances was caused between the 1960s and 1980s when at least 680 km^2 (16 %) of the Flow Country peatlands were ploughed and afforested with plantations of non-native conifers (Stroud *et al*. 1987). Standing trees on peatlands can change water quality in receiving watercourses by acidification, capture of elements, altered loadings of dissolved carbon, particulate carbon and other sediments, and by altering geomorphology and hydrology (Hornung *et al*. 1987, Miller *et al*. 1996, Gilvear *et al*. 2002). In turn, these changes may adversely affect salmon populations (Gilvear *et al*. 2002).

Amid concerns about the wider effects of afforestation on biodiversity and about the fate of the large store of carbon in the underlying peat (Cannell *et al*. 1993), the first large-scale forest-to-bog restoration initiatives of the Royal Society for the Protection of Birds (RSPB) took place in the late 1990s at the Forsinard Flows National Nature Reserve (NNR). By 2012, trees had been removed from more than 15 km^2 of the Flow Country and further drain blocking had been conducted across 150 km^2 in an attempt to restore functional blanket bog habitat. Evidence of the detrimental effect of afforestation on wader populations in adjacent open bog (Hancock *et al*. 2009, Wilson *et al*. 2014) also triggered key changes in policy. Forestry must now be removed to create buffer areas around designated peatland sites, and planting on peat > 40 cm deep is no longer permitted (FCS 2014). With the Scottish government pledging support to peatland restoration (SNH 2015), and with plantations reaching the end of their growth rotation period, forestry removal is likely to be a significant land-use change over the coming decade. During initial restoration trials, trees were small enough to be felled and pushed into the plough furrows. As the trees got bigger over time and with the prospect of using the trees for biomass energy production, new methods were needed; and whole tree harvesting, mulching and brashing requiring specialised machinery have been developed. Although blanket bog and freshwater habitats may eventually benefit from these land use changes, in the short term they are intensive interventions creating disturbance that could affect both the peatlands and adjacent interconnected ecosystems such as the freshwater streams and rivers which are crucially important for salmon.

A notable effect of forestry removal on water quality is the increased N and P concentrations and export related to decomposition of brash and needle litter left on site. This effect has been noted in both soil pore water (Asam *et al*. 2014, Gaffney *et al*. 2018) and stream water in catchments where restoration work has been undertaken (Rodgers *et al*. 2010, Asam *et al*. 2014, Finnegan *et al*. 2014, Kiikkilä *et al*. 2014, O'Driscoll *et al*. 2014a, Clarke *et al*. 2015, Smith 2016), but usually subsides after 2–4 years. In some cases increased Al and Fe concentrations and exports have also been observed (Müller & Tankere-Müller 2012) and related to changes in pH and organic matter, and to the chemical composition of the peat. The effects of forest felling on natural biogeochemical cycling and fluvial transport of sediments, carbon, nutrients and metals can in turn affect stream ecology (Neal *et al*. 2004, Ramchunder *et al*. 2009, Finnegan *et al*. 2014), which led O'Driscoll *et al*. (2014b) to suggest that buffer areas between forestry and rivers could be an option for reducing P load and potential harm to freshwater organisms. Therefore, it is necessary to understand the effects of forest management on stream habitats and apply this understanding in careful planning of any restoration work.

Many previous studies have targeted surface water at convenient sampling locations, but few have focused on changes in the chemistry of hyporheic water (in the zone beneath and just above the streambed) at known salmon spawning sites ('redds'), and especially at depths within the streambed where salmon eggs are usually deposited. Consequently, this study was set up to assess such effects on known Atlantic salmon (*Salmo salar*) redd sites resulting from tree harvesting, brash removal and drain blocking undertaken as part of a forest-to-bog restoration trial. More specifically, we aimed to measure: 1) the immediate effects (< 6 months) of management on water properties at different depths in the hyporheic zone; 2) the short-term (1 year) changes in water chemistry; and 3) sediment deposition at the redd sites upstream of and downstream from the sites undergoing restoration. We thought it likely that the immediate effects would be negligible because of dilution due to high precipitation during the main management period. In the short term we expected to find detectable increases in some metals downstream of the restoration site, resulting from needle decomposition. We also thought it likely that sediment load would be similar between sites and over time because of the small proportion of catchment affected and because existing sediment traps may buffer any potential effects.

METHODS

Site description and sampling methods

The study took place in the Dyke River, a tributary of the Halladale River, which is one of the major river systems in Sutherland. The Dyke River and the burns and streams that feed it are known to provide habitat for Atlantic salmon. The sub-catchment where the study took place covers approximately 40 km^2 comprising: 1) about 27 km^2 of open bog including about 2 km^2 which was drained in preparation for forestry but never afforested, the ditches now having been blocked; 2) 4 km^2 of formerly afforested blanket bog that underwent forest-to-bog restoration in 2003–2005 (10 %); and 3) 9 km^2 of non-native forestry of which about 1 km^2 (2.5 %) underwent restoration during the course of this study (Figure 1a). As part of the restoration management, the collector drains were blocked (November 2013 to January 2014) by installing plastic piling dams upstream of the point where the drain entered a watercourse, with silt traps made of Hytex Terrastop® installed across the drain upstream of the plastic piling to reduce sediment transport. From November 2014, a combination of standard felling (main stems removed and brash laid in mats on the surface for machinery to drive on) and enhanced felling (brash mat removed post-harvest) was used to remove the trees. This work was completed by April 2015.

In autumn 2013, the Dyke and its tributary burns were surveyed to identify salmon redd sites; and in autumn 2014, prior to spawning, ten of those sites along a short segment of the river were temporarily covered with stone slabs to prevent spawning. Four spawning sites (Figure 1, crosses) were located upstream of the forestry plantation in an open area

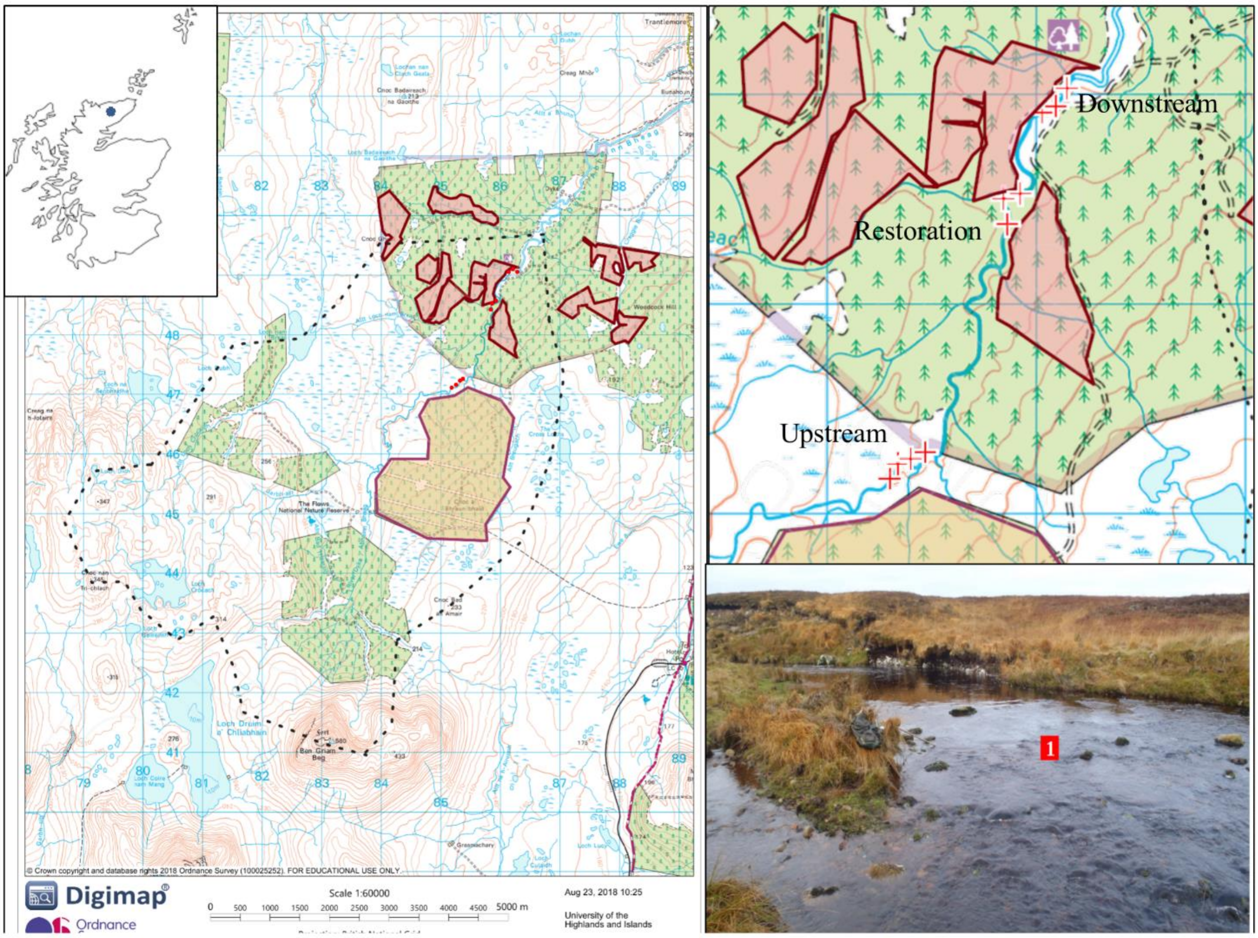

Figure 1. Details of the study location. Inset: blue star indicates location of the study in Scotland. a) Map of the sub-catchment (dotted black line indicates its boundary) located in the RSPB's Forsinard Flows NNR around the Dyke forest, which receives water from open bog, forestry plantations (green areas), older forest-to-bog restoration areas (restoration in 2005–2006; yellow-brown shading) and areas within the plantation undergoing restoration during the study (red shading). b) Locations, marked by crosses, of the sampling points along the Dyke River within the sub-catchment upstream ($n = 4$), within ($n = 3$) and downstream ($n = 3$) of the areas undergoing active forest-to-bog restoration management. c) Photograph of the first upstream sampling location (indicated by the red label) within the Dyke River.

comprising a mixture of forest-to-bog restoration (restored 2004–2006) and drained-blocked blanket bog ('upstream', U); three sites were within a section of the plantation where forest-to-bog restoration management was ongoing ('restoration', R); and three sites were downstream of the restored area, still within the plantation ('downstream', D). Hyporheic sampling tube constructed from 6 mm internal diameter (ID) polyethylene tubing, shielded with a fine mesh, was inserted in the streambed at depths of 75, 175 and 275 mm in each site and connected to the surface, enabling collection of water samples from each depth separately (Youngson *et al*. 2005, Malcolm *et al*. 2009). Samples (250 ml) from each depth and from the surface were collected periodically using a hand-held vacuum pump and clean polyethylene bottles between November 2014 and March 2015, covering the duration of salmon egg incubation and hatching. This also spanned the period during and immediately after restoration. At each sampling date, stage height (i.e. the height of the water surface above a locally established datum level) was measured; and pH, dissolved oxygen (DO) concentration and electrical conductivity (EC) were determined with manual probes on the four samples collected from each of the ten sites.

Three sets of 250 ml hyporheic water samples from the 75, 175 and 275 mm depths were also collected at the same ten sites, on 01 November 2014 (onset), 11 December 2014 (during intervention) and 16 April 2016 (one year after the end of the management). These samples were taken to the laboratory, where concentrations of several elements (K^+, Na^+, Ca^{2+}, Mg^{2+} Cu, Fe, Al, Mn, S and Zn) were determined by Inductively Coupled Plasma Optical Emission Spectrophotometry (ICP-OES) using a Varian 720 ES (Clesceri *et al*. 1989). Briefly, samples were diluted with Milli-Q water to reduce concentrations to within the linear range of the instrument. Standards were made from single element ICP standards (Fluka), made into a multi-element intermediate which was then diluted into five calibration standards. Samples with concentrations > 110 % of the top standard were further diluted and re-run until they fell within the calibration range. A certified reference material (Big Moose Lake Water; Environment Canada) was analysed during every instrument run for quality control purposes.

In January 2015, two pairs of baskets (230 mm square slatted plastic aquatic plant holders) were installed in the streambed to collect and measure sediment deposition at the upstream and downstream sites. The baskets were buried so their tops lay flush with the gravel surface. Because lifting, emptying and replacing the sediment baskets causes in-stream disturbance, sediments were collected only three times, on 02 and 23 February 2015 (during management) and 08 April 2015 (post management). At each removal, the accumulated sediments were taken back to the laboratory, dried, sieved to separate them into six different particle size categories (< 0.25, 0.25–0.5, 0.5–1.0, 1.0–2.0, 2.0–4.0 and 4.0–8.0 mm) and weighed. Loss on ignition (LOI) was measured from the 0.25–0.5, 1.0–2.0 and 2.0–4.0 mm fractions. The sediments were dried and placed in pre-weighed crucibles, weighed, placed in a muffle furnace for four hours at 550 ºC, removed, left to cool in a desiccator, and weighed again. Proportion by dry mass of organic matter in the sediments was determined as LOI divided by initial dry sediment mass. Rate of sediment deposition was calculated as the total weight (all fractions) of sediment deposited divided by the number of days the sediment basket had been in the gravel bed and the surface area of the basket.

Statistical analyses

R software (R Core Team 2017) was used for all of the statistical analyses. Since all of the water samples were collected from the same stretch of river, the data were potentially auto-correlated. This spatial auto-correlation can be modelled and taken into account in the subsequent comparison of sites. As a first step, an asymmetric eigenvector map (AEM) was constructed to create a matrix of spatial variables describing the spatial connectivity between sampling points as dictated by the direction of flow (upstream to downstream of the forest) (Blanchet *et al*. 2008). Redundancy analysis (RDA) was used to test whether any of the spatial variables generated were significantly related to pH, DO and EC. This indicated whether the chemistry of a particular sampling point was likely to be more similar to that at sampling points downstream because of the direction of flow, and how far the effect was likely to extend. Spatial connectivity between the ten sampling sites was not significantly related to the water chemistry variables measured ($F = 0.28$, $P = 0.99$). Therefore, all of the sampling sites were considered to be spatially independent in further analyses.

Redundancy analysis (RDA) was then used to assess whether stage height influenced the physico-chemical variables. As its effect was significant, we tested whether individual physico-chemical variables (pH, DO, EC) varied across depths within sites over time using stage height as a covariable. We performed linear mixed models (function nlm, package nlme) with site and depths within sites as fixed factors and sampling date as a random slope, which allows the variation in water chemistry over time to be different for each site. For each variable, F and P values were then generated with the function *anova*. Finally, Pearson's correlations were used to examine how variables related to each other.

For the element analyses, a full time series was not available. The effect of restoration is indistinguishable from an effect of seasonality in itself. Instead of using each time point individually, the difference in metal concentration between the post-management sampling and the mean of the two samples taken during intervention was calculated for each variable. We used the difference to test the effect of site (U, R, D) and depth within site (75, 175, 275 mm) with multivariate analysis of variance (function manova, package stats). In this case, an effect of restoration would appear as a significantly greater or smaller difference between sites. Where relevant, log transformations were used to improve homoscedasticity.

For the sediments, because of the small number of replicates ($n = 2$) it was not appropriate to conduct statistical analyses. Therefore, the distribution of sediment fractions, the total mass of sediments, LOI and organic matter (OM) were examined visually using simple graphical comparisons.

RESULTS

EC, DO and pH varied similarly over time in the three sets of sites, and the values were not significantly different between sites for a given depth (Figure 2). The values of EC were highest in January and February at all sites and depths, corresponding to high precipitation events (Figures 2 and 3). The pH values were inversely related to stage height

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(Pearson's $R = -0.88$), with higher pH at low stage heights and low pH at high stage heights, while no such trend was observed for EC (Pearson's $R = 0.22$).

Of all the elements analysed, only potassium and zinc were at significantly higher concentration in the restoration sites ($F = 12.5$, $P < 0.001$ for K^+, $F = 4.14$, $P = 0.02$ for Zn) after management was undertaken. Effects were more pronounced at the 75 and 175 cm depths, where concentrations of K^+ and Zn were almost twice as high post management than in the open bog upstream and the standing forestry downstream (Figure 4). No other significant changes in element concentrations were detected that could be attributed to restoration management. For Fe (75 mm), Mg^{2+} (75 and 175 mm) and Mn (all depths), concentrations were lower in the samples collected post management than in the samples collected during management at all of the sites

The proportion of organic matter was < 10 % in all sediments (Figures 5a and 5b). Overall, the upstream sites, which mostly drain open blanket bog and older restored sites, received more sediment than the downstream sites, which drain afforested and restoration sections (Figures 5c and 5d). The size class distributions of the sediment remained similar, with coarser sediments (4.0–8.0 mm) contributing about 30 % and 25 % of the overall sediment mass in the upstream and downstream sites, respectively. For a given site, the rate of sediment deposition was higher in the winter months when management was ongoing (375 ± 50 g d^{-1} m^{-2} upstream, 264 ± 78 g d^{-1} m^{-2} downstream) than during the following spring (227 ± 77 g d^{-1} m^{-2} upstream, 118 ± 9 g d^{-1} m^{-2} downstream).

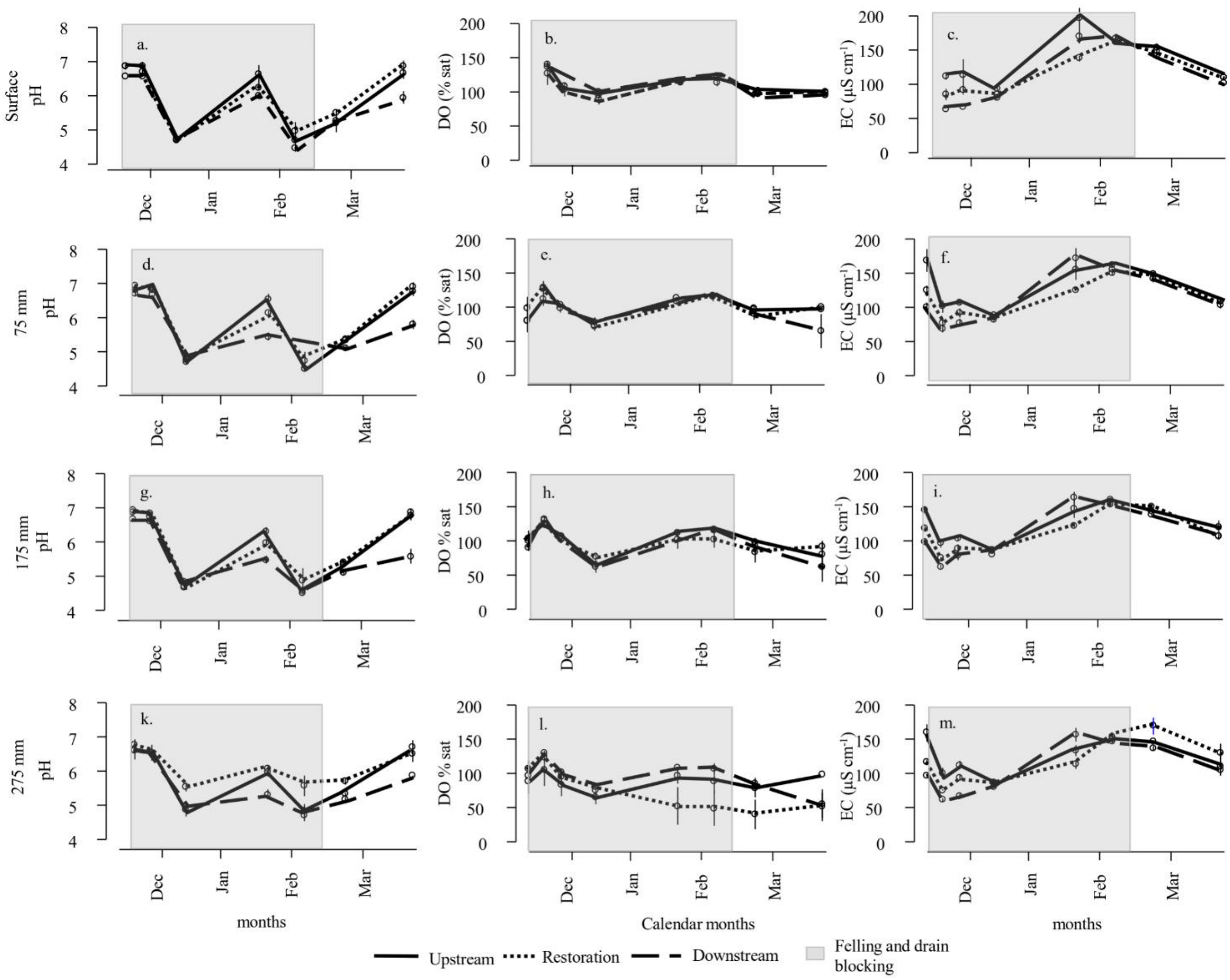

Figure 2. Temporal trends for pH, dissolved oxygen (DO) and electrical conductivity (EC) in water samples collected at the surface (a–c) and at depths of 75 mm (d–f), 175mm (g–i) and 275 mm (k–m) in the River Dyke at sites upstream of (black line), within (dotted line) and downstream of the forest undergoing restoration work (dashed line) between November 2014 and March 2015. Shaded areas indicate the period of active management (tree felling and drain blocking).

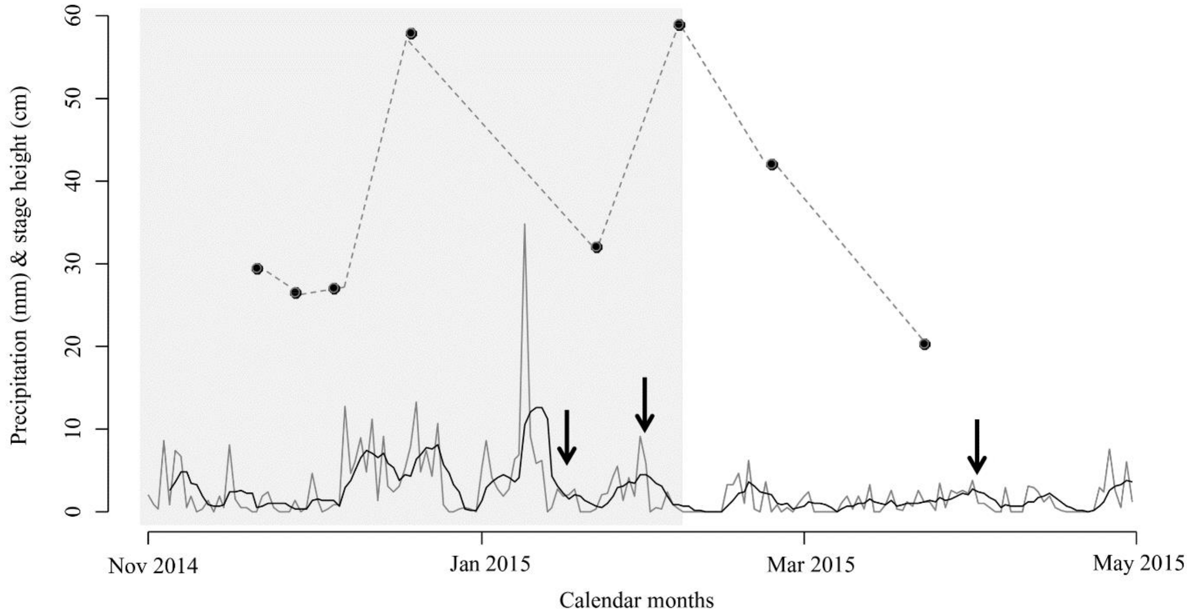

Figure 3. Stage height in cm (points and dashed line), daily precipitation in mm (solid grey line) and five-day average precipitation in mm (solid black line) during the sampling period November 2014 to May 2015, during which restoration management was undertaken (shaded area). The three sediment basket recovery dates are identified by the black arrows.

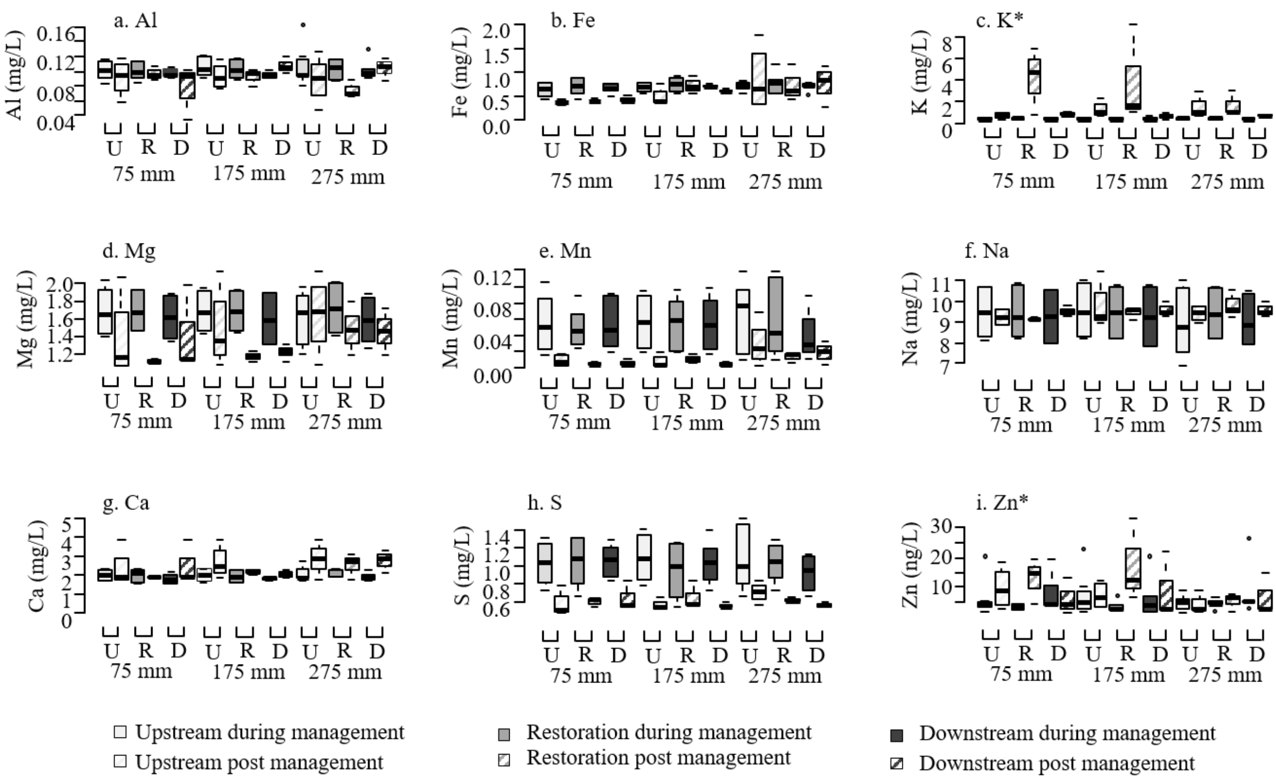

Figure 4. Concentration of elements (Al, Fe, K, Mg, Mn, Na, Ca, S, Zn) in the water taken at depths of 75, 175 and 275 mm in the streambed during (shaded boxes) and one year after restoration (hashed shaded boxes) upstream of (U, light grey), within (R, dark grey) and downstream of (D, black) the restoration management areas. Within U, R and D, bars are in pairs; the first is during management and the second after management. Elements followed by a * indicate a significant effect of restoration management. Note the change of units for Zn.

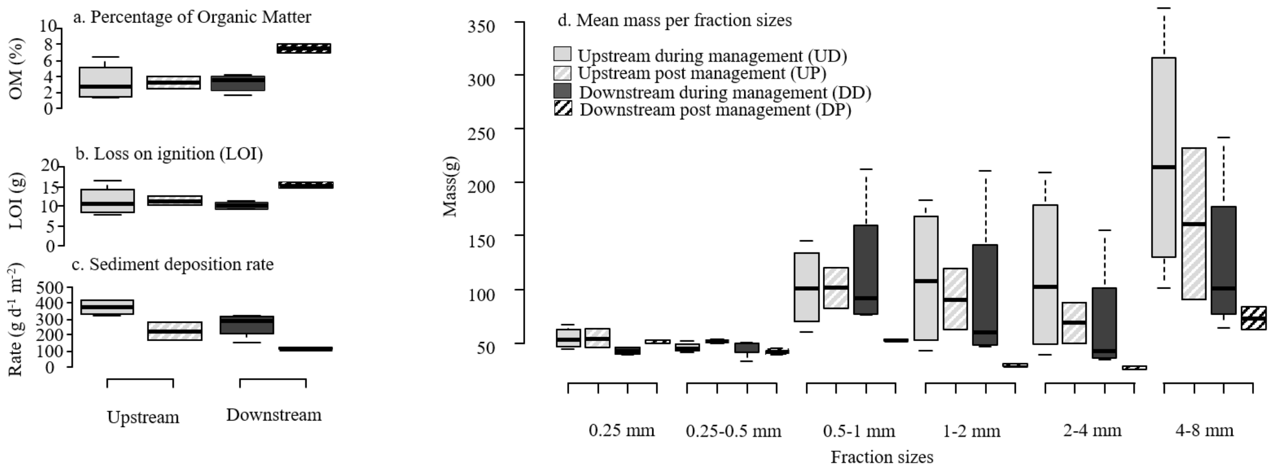

Figure 5. Distributions of a) OM (fractions 0–4 mm only), b) loss on ignition (LOI; fraction 0–4 mm only), c) sediment deposition rates and d) mean mass per fraction size in sediment baskets located in the Dyke river bed at sites upstream (light grey) and downstream (black) of restoration work. The two pairs of sediment baskets at each site were retrieved, emptied and replaced during (two collections) and after (one collection, hashed boxes) restoration management activities (see Figure 2).

DISCUSSION

Changes in water chemistry

A possible explanation for the lack of spatial autocorrelation in our sites is that each sampled stretch of river receives water not only from different subsidiary burns, but also directly as near-surface throughflow and overland flow, thereby diluting local effects. This is the case in many other UK rivers which drain areas of upland blanket peat (Holden & Burt 2003).

Our results suggest that restoration management (felling and drain blocking) had little or no effect on water quality (DO, EC, pH) in the short term (months). While it was not significant and rather variable, we noted a slight decrease in DO over time at depth (275 mm) in the sites undergoing restoration. Low DO concentration at depths where eggs may be buried could potentially damage incubating eggs and salmon in early stages of development (Alderdice *et al*. 1958, Hamor & Garside 1976, Malcolm *et al*. 2005). It could be linked to potential increases in solutes such as PO_4, which is known, first, to be leached from forest-to-bog sites (Renou-Wilson & Farrell 2007) including at our sampling location (Gaffney 2016) and, secondly, to cause concern for eutrophication and associated depletion of DO in streams draining upland peaty catchments (O'Driscoll *et al*. 2011). Similarly, increases in DOC concentration associated with storms (Clarke *et al*. 2015) or physical disturbance in peaty catchments (Müller & Tankere-Müller 2012) could lead to increased respiration in the hyporheic zone and DO depletion (Soulsby *et al*. 2001). Another explanation could be the variable contributions of surface water and long-residence groundwater of varying provenance in the hyporheic mixing zone. Thus, DO at 275 mm depth could fall with time as the site re-stabilises and upwelling deoxygenated groundwater becomes more dominant over downwelling surface water. Nevertheless, a sampling campaign across a wider range of hydrological conditions and a better characterisation of the Dyke's morphology would be needed to clearly assess the relative contributions of upwelling and downwelling at different times.

We did not detect any significant changes in elemental water chemistry, with the notable exceptions of K^+ and Zn which appeared to have risen especially at the shallower hyporheic depths sampled (75 mm and 175 mm within the gravel bed). Both elements are commonly leached from brash litter left on site after forestry removal (Palviainen *et al*. 2004, Fahey *et al*. 1991, Gaffney 2016, Gaffney *et al*. 2018). Despite the very small proportion of the catchment being felled, the sampling sites within the restoration area were < 100 m from the edges and immediately downstream of a small burn which runs through the areas where brash would have been laid out and left on the ground. Therefore, export of K^+ and Zn through overland flow, seepage and via the burn to the Dyke, coupled with downwelling, could explain the raised values. However, high precipitation and consequent high volume of surface runoff could have contributed to reducing any detectable effects at the redd sites farther downstream through dilution (Proctor 2006). Potassium is an essential macronutrient not known to have a toxic effect on salmon. Zinc, on the other hand, can be toxic to salmonids and the toxicity is known to vary between life stages, with newly hatched alevins more resistant than older juveniles (Chapman 1978). Despite the significant effect of restoration management, the concentrations of Zn measured are all less than 30 ng L^{-1} and therefore within known safe levels (at or below 1 μg L^{-1}; Chapman 1978) and well below published LC_{50} concentrations for Atlantic salmon (0.35–1.60 mg L^{-1}; Farmer & Ashfield 1979).

While more frequent sampling and, in particular, sampling across a range of hydrological conditions might have been desirable, similar results have been obtained in the Halladale River, into which the Dyke River flows (N. Shah, personal communication) and in the Thurso River (Müller *et al*. 2015). These lend support to our results. A longer-term study conducted in first-order streams feeding the Dyke River and, farther on, the Halladale River detected increases in concentrations of Al, Mn and PO_4 as well as K^+ which started during the summer following the restoration period, but only where the increases coincided with low precipitation and high temperature (Gaffney 2016). Thus, our results seem consistent with the general idea that, in blanket bog micro-catchments, the presence of undisturbed areas buffers against potential changes in the water chemistry of receiving watercourses caused by restoration management. Although we deliberately focused on areas where we know that salmon spawn in order to fill a gap in our understanding of direct effects of forest-to-bog restoration on salmon habitat, further studies could include burns and streams that feed the Dyke to help us understand the mechanisms better.

We observed an inverse relationship between pH and stage height at all of our sites. Low water level in the Dyke River equals low flow, during which acidity can be neutralised by calcium carbonate ($CaCO_3$) in streambed sediments (Miller *et al*. 2001). A separate study in the same river has often measured seasonal increases in acid-neutralising capacity associated with low flows in summer months (Gaffney 2016).

Sediment load

Some studies have shown increases in sediment load and water runoff following conifer afforestation or logging; a consequence of mechanical causes (e.g. drainage, ploughing, vehicle movement) and physical alterations (e.g. cracking, drying) of the peat mass and the mineral soil underneath (Ratcliffe & Thompson 1988, Platts *et al*. 1989, Ramchunder *et al*. 2009, Clarke *et al*. 2015). In the present study, the mass and rates of the sediment deposited in the baskets appeared more influenced by the season and inherent catchment characteristics than by management. The apparent decrease between the "during restoration" and "post restoration" periods might be an artefact: there were relatively heavy (> 10 mm) and sustained precipitation events likely to bring sediments in winter when the first two recoveries of sediment baskets were made, and much less precipitation, associated with lower stage height and flow prior to the spring collection.

It is interesting to note that sediment accretion appears to decrease more in the downstream site. This may be a consequence of drain blocking and silt traps installed along the collector drains as part of the restoration management. They were put in place to mitigate any potential export of sediments resulting from the work of the machinery and the physical disturbance. In this particular case, at least two sediment traps in each drainage channel were used, as the second (or third) ones are an important back up to the first during big rainfall events, or as a result of the upper one filling up. A geotextile screen was installed downstream of the sumps to capture any sediments. The screens are porous, but trap all but fine sediment moving downstream. Retaining fine sediments is essential to avoid infiltration, DO depletion and reduced permeability of spawning gravel (Soulsby *et al*. 2001) and while this mitigation measure should be encouraged wherever peatland management is taking place, further work could explore which of the various combinations of blocking, traps, screens and dams is most effective at minimising export of particulates of all sizes across a range of weather conditions.

Limitations of the study and future work

This study was limited over time and space, did not include replication at the stream level and did not include N or P. The main reasons for this were: (1) there were no other locations within the nature reserve where the study took place for which spawning sites happened to be located upstream, within and downstream of restoration on a continuous stretch of river; (2) there were more spawning sites available along the Dyke but we wanted to limit damage to the salmon and avoid utilising more spawning sites for research purposes than needed in the first instance; and (3) while N and P would have been interesting to assess, they were not our focus and would have required a different sampling strategy and processing.

Our study demonstrates that there were no immediate effects of management at known spawning sites in the river Dyke, on sediment load on the river bed, nor on EC, DO, pH and at least not on all but two of the measured elements. The notable exceptions were potassium and zinc. Those elements, perhaps alongside N, P and possibly other known toxic elements to salmonids such as cadmiun and copper (Chapman 1978) should be integrated into future monitoring. Further, while we report here only total Al concentrations, we know that Al has a complex chemistry influenced among other things by pH. It is toxic to salmonids only in its inorganic free form (Al^{3+}) (Kroglund & Staurnes 1999), so in the future we recommend that Al species might be important to target for assessing effects on salmonids. We noted that, compared to open and older restored bog, forested areas could be associated with higher sediment export but sediment traps appear effective at reducing the load resulting from machinery-related disturbances. Further work would still be required to fully understand how effective they are. We also observed an influence of sampling date associated with stage height on the physico-chemistry of the river, especially for Fe, Mg and Mn, which all had lower concentrations in the sampling post-restoration (lowest stage height) across all sites. If sampling had targeted only restoration sites without upstream controls, our results might have been misinterpreted as an effect of restoration. This clearly emphasises the importance of context and reference sites when evaluating land-use change.

We are aware that the sampling period was short and that some of the effects, such as solute exports, may start to appear only after two years or more, as was seen on the pore water quality in the peat and in receiving watercourses at the same study site (Gaffney *et al*. 2018, Gaffney 2016). We therefore strongly advocate continued monitoring to keep documenting the effects of forest-to-bog management on freshwater systems. We acknowledge that our current results, interpretation and conclusions may be relatively limited in their wider application, but we believe that our pilot study is a first step towards understanding the effects of forest-to-bog restoration on freshwater habitats in Scotland.

Short-term changes associated with management are likely to be catchment-specific to some degree and influenced by local climatic and weather conditions, in particular the intensity and timings of droughts and storms. From a freshwater biodiversity perspective, management should aim to minimise and mitigate short-term pulses in solute exports, for example by removing brash (the main source of P and Zn and possibly other elements), installing and testing the efficacy of sediment traps in larger drains, creating buffer areas, etc. In any case, it may be argued that short-lived ecosystem changes resulting from disturbance and improved water quality arising from restoration may still be better than longer-term chemical and physical alteration to watercourses caused by continued forestry plantations. Further research should aim to assess the direct responses of freshwater organisms to forest-to-bog restoration management, and understand the hydrological processes driving the chemical changes at a wider catchment scale and over longer time periods.

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AUTHOR CONTRIBUTIONS

RA, NC and AY conceived the study; RT collected samples and conducted all the fieldwork; DS completed all the laboratory work; RT and DS compiled the data; RA processed the data, made statistical analyses and wrote the manuscript, with inputs from NC and AY.

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