

The Tarland Catchment Initiative and Its Effect on Stream Water Quality and Macroinvertebrate Indices

J. Bergfur, B. O. L. Demars,* M. I. Stutter, S. J. Langan, and N. Friberg

The Tarland Catchment Initiative is a partnership venture between researchers, land managers, regulators, and the local community. Its aims are to improve water quality, promote biodiversity, and increase awareness of catchment management. In this study, the effects of buffer strip installations and remediation of a large septic tank effluent were appraised by water physico-chemistry (suspended solids, NO_3 , NH_4 , soluble reactive P) and stream macroinvertebrate indices used by the Scottish Environmental Protection Agency. It was done during before and after interventions over an 8-yr period using a paired catchment approach. Because macroinvertebrate indices were previously shown to respond negatively to suspended solid concentrations in the study area, the installation of buffer strips along the headwaters was expected to improve macroinvertebrate scores. Although water quality (soluble reactive P, NH_4) improved downstream of the septic tank effluent after remediation, there was no detectable change in macroinvertebrate scores. Buffer strip installations in the headwaters had no measurable effects (beyond possible weak trends) on water quality or macroinvertebrate scores. Either the buffer strips have so far been ineffective or ineffectiveness of assessment methods and sampling frequency and time lags in recovery prevent us detecting reliable effects. To explain and appreciate these constraints on measuring stream recovery, continuous capacity building with land managers and other stakeholders is essential; otherwise, the feasibility of undertaking sufficient management interventions is likely to be compromised and projects deemed unsuccessful.

THE TARLAND CATCHMENT INITIATIVE has deployed large joint efforts among land managers, regulatory and conservation agencies, and scientists through a participative process. Its aims are to curb diffuse and point-source pollution through buffer strips, septic tank improvements, and the diversion of wastewater treatment plant effluent (Stutter et al., 2010) and to increase habitat diversity through the creation of wetlands and channel restoration. Integrated and holistic catchment management has long been promoted in the scientific literature, particularly regarding the key processes required for successful river restoration (e.g., Harper et al., 1999; Ormerod, 2004; Palmer et al., 2005; Zalewski et al., 2008). This is now largely driven by policy and legislation, such as the Water Framework Directive (WFD, 2000/60/EC) in Europe, with its focus on ecological status and stakeholder engagement.

To reduce the impact of diffuse pollution from agricultural land management on the environment, a range of best management practices (BMPs) (Cuttle et al., 2007) have been included as central to government support to farm agri-environment schemes and good codes of practice (SEERAD, 2005). Many of these BMPs have not been evaluated in the United Kingdom (Kay et al., 2009), particularly regarding their effects on aquatic and riparian wildlife (Hilton, 2002) at the catchment scale. River ecology responds to natural processes across a range of spatial and temporal scales and multiple anthropogenic stressors (Ranganath et al., 2009; Death and Collier, 2010; Friberg, 2010a), so it is often difficult to disentangle the multiple drivers outside an experimental setting (Townsend et al., 2008; Matthaei et al., 2010). Moreover, the time lag in water quality, bed sediment, and ecological response to BMPs can be long (Meals et al., 2010) and may require additional triggers, such as high-magnitude flow events (Benstead et al., 2007).

The lack of consideration of river processes and spatial and temporal scales, together with study design (i.e., lack of pre-, post-, and reference location data), are key problems with most river restoration projects (e.g., Malakoff, 2004; Ormerod, 2004; Bernhardt et al., 2005). Hence, the current detection of ecological improvement after river restoration appears to be equivocal

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Abbreviations: BMP, best management practice; BMWP, British Monitoring Working Party; SRP, soluble reactive phosphorus; WFD, Water Framework Directive.

(Palmer et al., 2005; Alexander and Allan, 2007; Palmer et al., 2010). More robust studies are needed (Clews and Ormerod, 2010), as exemplified by the studies of Bishop et al. (2005), showing significant P load reduction due to BMPs; Benstead et al. (2007), showing stream recovery from nutrient addition experiments; and Ormerod and Durance (2009), showing stream recovery from acidification. New studies should not only incorporate an understanding of the biophysical processes but also take on the challenge to integrate the social, ethical, and economic dimensions of environmental management (Lanckester et al., 2009; Spash et al., 2009; Friberg, 2010b; Palmer et al., 2010).

Previous observations in our study system (mixed agriculture) showed that catchment percentage area of intensive grassland correlated well with stream suspended solid concentrations (sediment supply), which in turn was negatively correlated to stream macroinvertebrate indices (Stutter et al., 2007). When fine sediment supply from riparian and catchment modifications exceeds in-stream transport and particle sorting capacity, the river bed becomes clogged (Schalchli, 1992), and this affects the web of aquatic wildlife, including macroinvertebrates (Wood and Armitage, 1997; Arthington et al., 2010; Larsen et al., 2011 and references therein). Siltation (sediment accrual) affects rivers globally and, in the United States, is the principal source of impairment on the basis of stream distance affected (USEPA, 2000).

This paper reports on the restoration of headwaters through management intervention to provide buffer strips and better private waste water treatment, for which data have been collected (similarly to regulatory agency) using a before/after/control/intervention approach over an 8-yr period. More specifically, (i) we report the benefits of raising awareness and stakeholder participation in relation to BMPs and the need for participative catchment management, and (ii) we test whether restoration of individual headwaters has had measurable effects on physico-chemistry (suspended solids, nutrients) and ecology (macroinvertebrate indices). We hypothesize that a septic tank improvement and installation of buffer strips should decrease nutrients (NO_3 , NH_4 , PO_4) and sediment loadings in surface water, which in turn should improve macroinvertebrate indices. We anticipate that water physico-chemistry will respond faster than the macroinvertebrate community (Parkyn et al., 2003; Benstead et al., 2007).

Materials and Methods

Study Catchment

The River Dee in Aberdeenshire is one of the major river systems in Scotland. It is renowned for its important population of Atlantic salmon (*Salmo salar*), freshwater pearl mussels (*Margaritifera margaritifera*), and otter (*Lutra lutra*). The conservational importance of the river for these species has resulted in the main stem and its tributaries being designated a Special Area of Conservation under the provisions of the European Habitats Directive.

Within the River Dee, the Tarland Burn catchment (70 km²) is the most upstream tributary that is dominated by intensive land use. Langan et al. (1997) showed the significant input this had on the downstream water quality of the main stem of the River Dee. The Tarland catchment land use (Fig. 1) is typical for many agricultural regions of Northeast Scotland, in which the major land uses are arable (25%), plantation forestry (19%), improved and unimproved grassland (36

and 10%, respectively), heather moorland (8%), and mixed/broadleaved woodland (2%). The only settlement is the village of Tarland (~650 inhabitants). Significant diffuse and point-source pollution issues have previously been reported related to fecal indicator organisms, suspended solids, and N and P losses (SEPA, 2000; Cooper et al., 2006; Stutter et al., 2010).

Tarland Catchment Initiative

Stakeholders have been involved in the establishment and prioritization of intervention works according to their interest and stake in the catchment. Farmers and practitioners were informed via formal and informal meetings about catchment management, environmental issues facing the catchment, and the need for intervention. To get these messages across and to learn from land managers and farmers' experiences, a number of presentations have been used that are based on visual graphical display of information that could be readily understood and discussed with land managers (see www.macaulay.ac.uk/tarland/). A steering group resulted from these meetings formed by the principal land managers (the MacRobert Trust) together with Agency and research staff (Scottish Environmental Protection Agency, Scottish Natural Heritage, the Macaulay Institute, and Aberdeenshire Council). This group has taken the available information and views expressed by the stakeholders, together with their individual expertise and through consensus agreed on the priority, scale, and type of interventions possible. For all the individuals and organizations involved, there has been a need to improve the level of understanding of how a catchment operates and to highlight some of the pressures and constraints. This increased capacity to understand has given rise to a greater awareness and involvement. As part of this process, the steering group had to modify the scope of the interventions to try to ensure they met with organizational objectives. For example, some buffer strips were widened to incorporate a community footpath network, which in turn provided increased awareness. These transactions have resulted in a greater willingness to undertake intervention management aimed at improving water quality and habitat diversity (Fig. 1 and 2). The objectives of the intervention have been to reduce inputs of diffuse pollution from livestock and arable (cereal) production to improve water quality and the ecological status of the riverine environment in a systematic, tributary-based approach.

Stream Water Quality

Water chemistry samples were collected from spatially nested sites ranging from <1 to 50 km² at Coull near the catchment outlet (Fig. 1). Samples from the sites were collected monthly to seasonally (~80 samples per site) as spot samples on the same day. The analytical protocols were the same throughout the study period as previously reported (e.g., Stutter et al., 2007). After collection, water samples were stored at 5°C in the dark, filtered (<0.45 μm) (Millipore, Sigma, UK) within 48 h, and generally analyzed within 72 h of collection. Suspended solids were determined gravimetrically on the filter. The filtrate was analyzed for conductivity, pH, and colorimetrically for nitrogen (N) and phosphorus (P) nutrient forms from 2000 to 2002 by Trax (Bran and Luebbe, Germany) and then by Skalar San++ (Skalar, Breda, The Netherlands). There was a change in the detection limits for soluble reactive P (SRP) between the Trax (0.002 mg L⁻¹) and the Skalar instruments (0.001 mg L⁻¹). However, because

so few samples (~3%) were at these limits, there was little bias introduced. Other detection limits were constant throughout the analyses: 0.005 mg L⁻¹ ammonium N (NH₄-N) and 0.10 mg L⁻¹ nitrate N (NO₃-N). Samples at or below the concentration detection limits (DL) were set to equal DL/2.

Stream Macroinvertebrates

Macroinvertebrates are widely used in biomonitoring in the United Kingdom and elsewhere (Wright et al., 2000) and are one of the key biological elements in the WFD. Therefore, macroinvertebrates were sampled from all sites using the

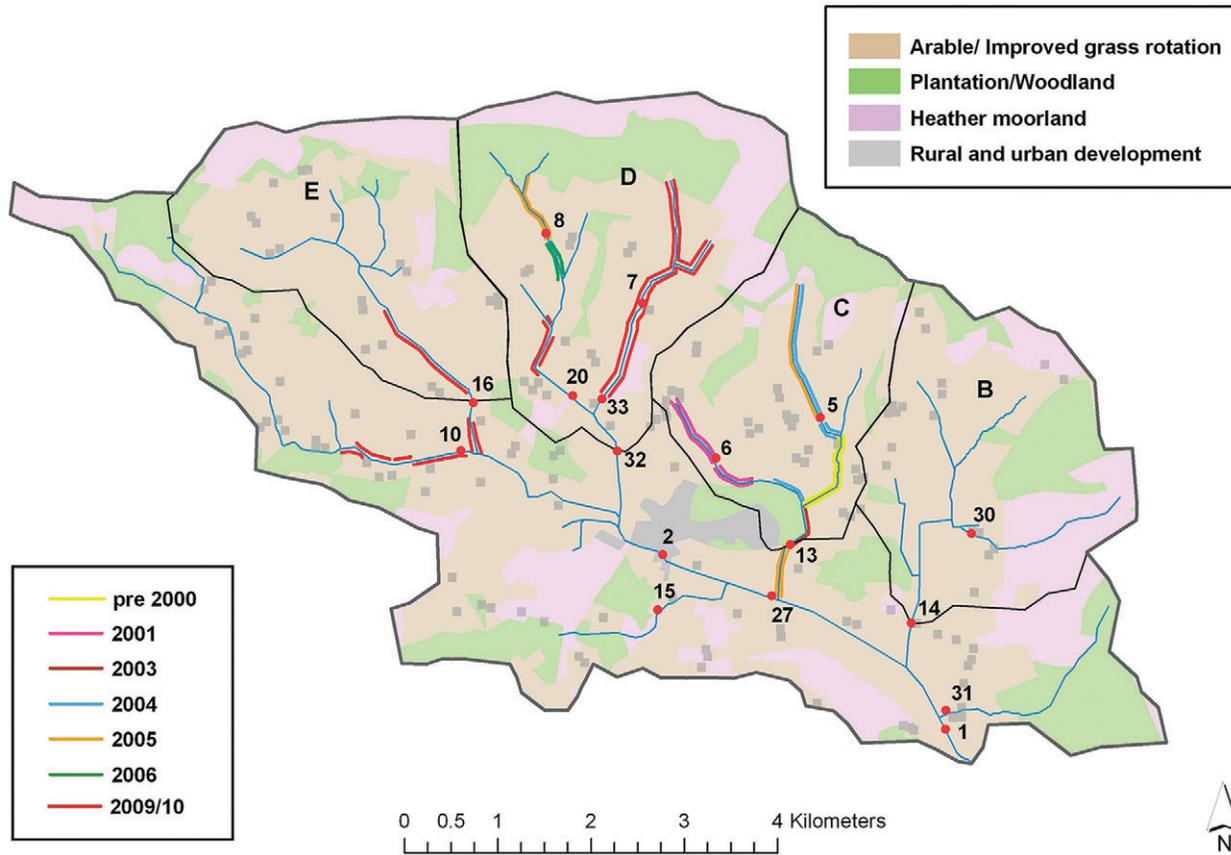


Fig. 1. Tarland catchment land use with date of buffer strip installation. Letters indicate tributaries, and numbers indicate stream sampling points. All restoration was completed before spring sampling that year.



Fig. 2. Photos of buffer strips before and after intervention. Top left, no riparian vegetation downstream from site 16; top right, left bank with buffer strip downstream from site 6; bottom left, site 13 before restoration; and bottom right, site 13 after restoration (wider buffer strip with footpath, plantation of native deciduous trees, cattle access to water transformed into a fenced wetland).

WFD-compliant RIVPACS sampling technique (Wright et al., 1984; Environment Agency, 1999) involving a habitat-proportional, 3-min kick sample using a 1-mm mesh hand net. Macroinvertebrates were sampled at the same 50-m reaches two to four times a year from September 2000 until January 2008 (≈ 20 samples per site). Samples were not taken at some sites on all dates due to weather conditions (drought, floods, or ice). In the laboratory, macroinvertebrates were identified to family level and assigned to four abundance classes: 1 to 9 individuals, class A; 10 to 99 individuals, class B; 100 to 999 individuals, class C; 1000 to 9999 individuals, class D. From these data, the British Monitoring Working Party (BMWP; sum of indicator taxa scores) (Wright et al., 1993; Hawkes, 1998) and Average Score per Taxa (BMWP divided by number of scoring taxa) (Armitage et al., 1983) were calculated. The total number of families, the number of EPT (*Ephemeropteran*, *Plecopteran*, and *Trichopteran*) families, and the proportion of insect families to the total number of families were calculated.

Study Design

The systematic and incremental restoration of the Tarland catchment was designed on a tributary basis starting with the most arable subcatchments (tributary C; Fig. 1). Some tributaries were kept as control. In total, 16 sites have been monitored over the past 10 yr. Here, we focus on four pairs of sites situated in the headwaters of the Tarland catchment for which we had substantial pre- and postappraisal monitoring data at comparable paired restored and control sites. Two restored sites (sites 5 and 8) have the same control (site 7), and one restored site (site 13) was tested against two different controls (sites 14 and 16). Further information about these sites is presented in Table 1, Fig. 1, and Supplemental Table S1.

Statistical Analyses

Restoration effects were tested with the random intervention analysis (Carpenter et al., 1989) using Canoco 4.5 (ter Braak and Smilauer, 2002) with 999 Monte Carlo random permutations

restricted for temporal structure (samples permuted using cyclic shifts using the same permutation in each site). An example of how the raw data were prepared for statistical analyses is provided in Supplemental Table S2. The effects of the predictors on the response variables were tested after removing the effects of covariables as indicated for each analysis performed in Supplemental Tables S3 and S4. The four pairs of sites provided replication (see Hurlbert, 1984). The advantage of this design is that the test is largely independent of other effects that may confound the results (e.g., seasonality). In this study there were no hydrological differences between pre- and postrestoration (Supplemental Fig. S1). Suspended solid data were $\ln(x + 1)$ transformed before statistical analyses to normalize the data and reduce heteroscedasticity.

The results are best visualized by plotting the difference between sites in physicochemical and ecological indicators against time and marking on the graph the time at which intervention happened (septic tanks) or was completed (sometimes buffer strip restoration took several years; see Fig. 1).

Results

The detailed statistical results (i.e., r^2 , F-ratio, P value, and magnitude of impact relative to overall average) are reported in Supplemental Tables S3 and S4.

Stream Water Quality

The strongest effect of the buffer strip intervention was a 5 and 10 $\mu\text{g L}^{-1}$ relative decrease in SRP and NH_4 , respectively, at restored site 5 after restoration relative to control site 7 (Fig. 3), although it was not statistically significant ($P \approx 0.1$) (Supplemental Table S2). This represented $\sim 40\%$ relative decrease in pollutant concentrations, which is substantial at those relatively low concentrations of NH_4 and SRP ($30 \pm 4 \mu\text{g N L}^{-1}$ and $16 \pm 3 \mu\text{g P L}^{-1}$, respectively, at site 5 before restoration). Generally, the buffer strips have had no significant effects ($P > 0.15$) on the four physicochemical determinants considered in this study (suspended solids, NO_3 , NH_4 , and SRP concentrations). The only significant ($P = 0.01$) intervention effect detected, from the removal

Table 1. Selected site characteristics in the Tarland catchment.

Tributary	Site	Area	Stream length	Mean slope	Intensively managed†	Plantation/ woodland‡	Extensively managed§	Restored/nature of intervention
		km^2	km	degrees		%		
B	14	7.4	7.8	9	55	19	25	control: degraded site, no remnant riparian vegetation
C	5	1.5	1.5	8	53	30	17	intervention: upstream buffered (fenced) and broadleaf trees planted (2004–2005)
	13	6.0	5.9	7	71	21	7	intervention: replacement of large septic tank and direct connection to stream removed (2002); installation of wetland and continuous buffer (both fenced), native woodland trees planted (1999–2005)
D	7	3.2	2.3	11	47	9	44	control: degraded site, some remnant riparian vegetation
	8	1.2	1.8	11	27	48	25	intervention: upstream buffered (fenced in 2005)
E	16	7.2	5.8	8	70	12	18	control: degraded tributary, some remnant riparian vegetation

† Arable and improved grassland rotation.

‡ Coniferous plantation and mixed woodland.

§ Rough grassland and heather moorland.

of a septic tank upstream of site 13, was on SRP and NH_4 concentrations (Fig. 4). This effect was significant, independently of the control sites used for the statistical analysis (sites 14 or 16). The effect of the septic tank improvement on NO_3 concentration was not as strong as for NH_4 and SRP and was only significant against control site 14; hence, it was probably not reliable (this may arise by chance due to the high number of tests performed).

Stream Macroinvertebrates

There were generally no intervention effects on the macroinvertebrate indices investigated. There was only one possible improvement detected at site 13, at which the BMWP score compared with the control site (site 14) showed an average linear increase from -22 to $+23$ over the 8-yr period. This may be the combined result from septic tank removal and buffer strip effects ($P \approx 0.1$) (Fig. 5 and Supplemental Table S4). Testing for individual effects using only part of the time series could not identify the relative role of the septic tank from the buffer strip effect (Supplemental Table S4). However, such improvement was not confirmed after using a different control site (site 16; $P > 0.5$) (Fig. 5).

The total number of families and the number of EPT (*Ephemeropteran*, *Plecopteran*, and *Trichopteran*) families were highly correlated to BMWP scores ($r > 0.9$; $n = 126$; $P < 0.001$) and effectively gave the same results. The relative abundance of insect families to total number of families was constant over time and across sites at 0.86 ± 0.06 (average ± 1 SD).

Discussion

Restoration Outcome

Other detailed studies have detected the impact of cattle, particularly at points of access to water (e.g., Owens et al., 1996; Davies-Colley et al., 2004; Miller et al., 2010a), although success in restoration efforts may be limited to certain environmental variables (Miller et al., 2010b). The present study reports findings from an extensive investigation designed at the catchment scale. This spatial extent is necessary for ecological improvement because short stretches of riparian management have not been found to benefit benthic stream macroinvertebrates (e.g., Ranganath et al., 2009; Death and Collier, 2010). One of the issues with small headwaters is perhaps a higher spatial and temporal variability due to, for example, rotation of land use activities (animal or crops), extent of soil drainage, and quality of riparian management before restoration. This variability makes it harder to detect change over time and may explain the different outcomes between a restored site against different control sites (see high between-site variability in Parkyn et al., 2003; Death and Collier, 2010). This limited evidence of water quality improvements might have been the product of inadequate monthly to seasonal sampling frequency in the studied headwa-

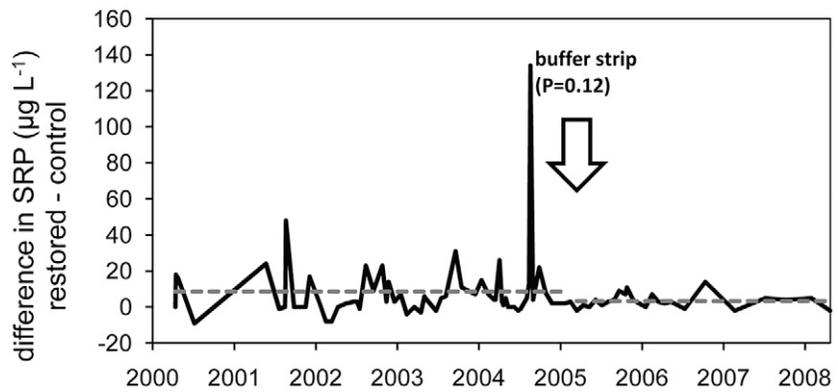


Fig. 3. Temporal differences (restored site 5 – control site 7) in stream water soluble reactive phosphorus (SRP) before and after intervention (arrow). Probability of intervention effectiveness was derived from random intervention analysis. The horizontal dashed gray lines indicate the average change in SRP difference before and after intervention.

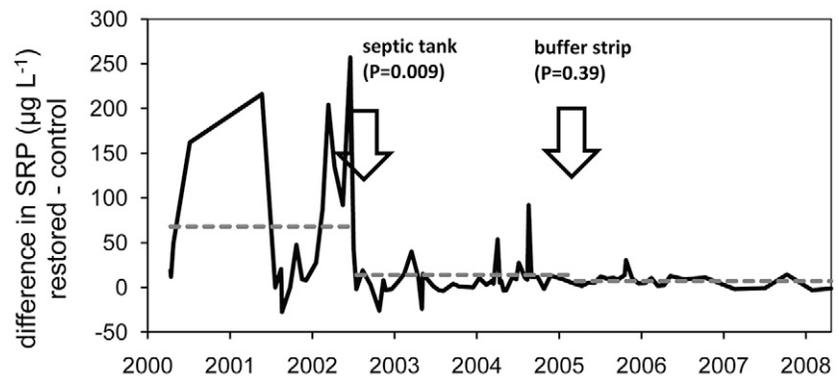


Fig. 4. Temporal differences (restored site 13 – control site 16) in stream water soluble reactive phosphorus (SRP) before and after interventions. Probabilities of individual intervention effectiveness were derived from random intervention analysis and variance partitioning analyses. The dashed gray lines indicate the average change in SRP difference before and after interventions. The raw SRP data are presented in Supplemental Fig. S2.

ters. Spot samples, as used in this study (and often the only data available from regulatory agencies), are invariably biased toward low flow conditions, whereas most of the suspended sediment and total P transfers occur mostly under high flows (Stutter et al., 2008). This calls for the use of semicontinuous monitoring technology such as turbidity probes to get better estimates and increase the chance of detecting a given improvement in water quality (Vinten et al., 2010). Another complementary method is the use of sediment traps collected at regular intervals to quantify fine sediment deposition.

Although there was no detected improvement in water quality due to buffer strips, macroinvertebrates could have responded to restoration measures because they integrate a wider range of conditions, such as changes in river bed sediment deposition (Larsen and Ormerod, 2010a; Larsen et al., 2011). It is likely that a longer period is required for macroinvertebrate scores to improve with the full development of riparian tree cover (Parkyn et al., 2003). Expectations in ecological improvement should also be related to the magnitude of ecosystem degradation and potential rehabilitation (environmental gradient) and statistical power of the analyses. The relatively low number of data points in the random intervention analysis may only be able to detect extremely large changes, and this is only likely to happen with very large environmental improvement. Although the role of

buffer strips in the mitigation of diffuse pollution may not always be effective due to preferential flow paths (natural or artificial; e.g., field drains), restoration of riparian habitats may start reconnecting the interdependence of stream–riparian ecosystems (Nakano and Murakami, 2001).

Riparian habitats also potentially provide a wide range of ecosystem services (Sweeney et al., 2004). Additionally, by providing a physical barrier between agricultural activities and the stream, buffers may stop certain types of pollution, such as agrochemical spray drift (Kay et al., 2009). However, buffers have uncertainties in their processes in relation to the cycling of nutrients. For example, Stutter et al. (2009) reported increased P leaching from riparian buffer strips relative to an unbuffered adjacent field under low flow conditions.

Although septic tank removal significantly decreased SRP and NH_4 concentrations, it did not seem to affect macroinvertebrates. A more direct pressure on macroinvertebrates is the partial pressure of oxygen in the water (Friberg et al., 2010). These measurements started later in the restoration program and could not be used in our analyses. Although the macroinvertebrates were sampled (site 13) 300 m downstream of the septic tank effluent, near complete re-aeration (95%) required a stream distance of about 4 km under low flow conditions (based on propane tracer study at site 13). In this instance, it is likely that dilution of the effluent was sufficient because the scores did not indicate gross organic pollution according to Scottish Environmental Protection Agency standards (UK TAG, 2008). Indeed, a similar macroinvertebrate index based on family level identification was related to septic tank density (weighted by overland flow path attenuation) in Australia (Walsh and Kunapo, 2009).

A refined survey (e.g., Larsen et al., 2009) or different types of indicators (see Friberg, 2010a; Friberg et al., 2010; Larsen and Ormerod, 2010a) may reveal a different outcome, such as the number of EPT species (rather than families) or species traits (Larsen and Ormerod, 2010b). Hence, it may be that the current low-cost monitoring, using similar data produced by regulatory agencies, is insufficient to appraise sediment problems and their remediation. The macroinvertebrate samples are archived, and it will therefore be possible to test other macroinvertebrate indices in the future, although improved taxonomic resolution will require additional identification skills and increased costs. Although macroinvertebrate abundance was not used in the present study, it should rely on counts rather than abundance classes, in line with what the regulatory agencies are now doing.

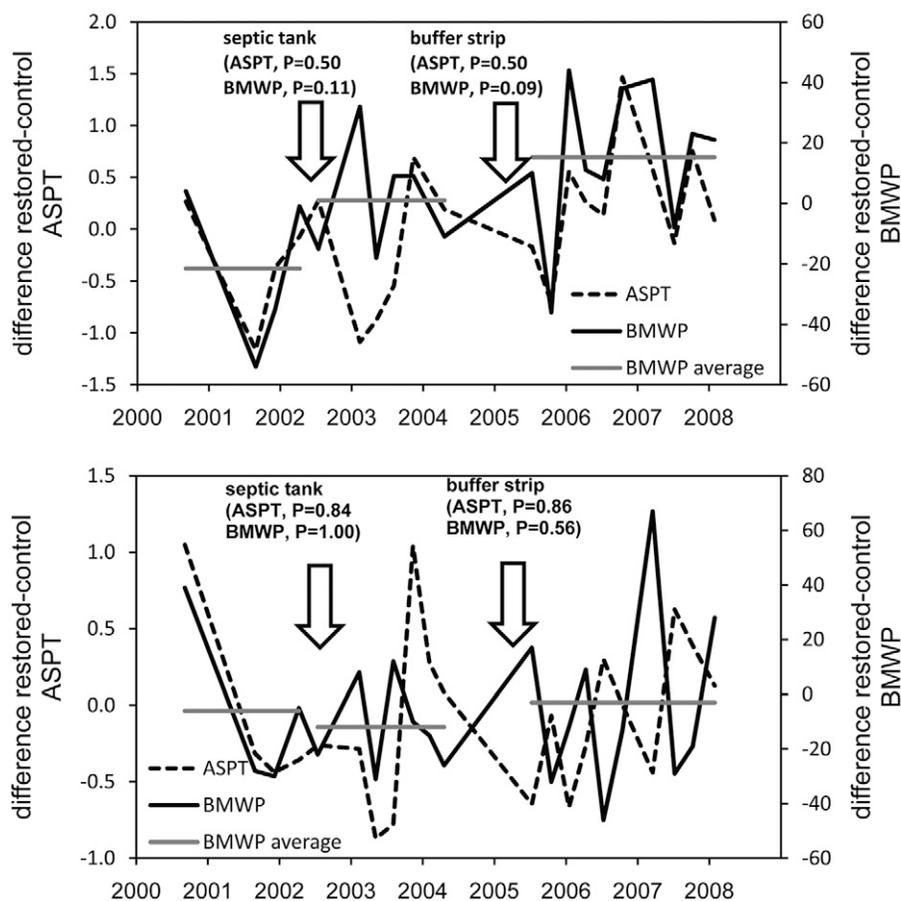


Fig. 5. Temporal differences (restored – control) in stream macroinvertebrate indices (Average Score per Taxa [ASPT], British Monitoring Working Party [BMWP]) before and after interventions. Restored site 13 against two different controls: top panel, site 14; bottom panel, site 16. Probabilities of individual intervention effectiveness were derived from Random Intervention Analysis (RIA) and variance partitioning analyses. The horizontal gray lines indicate the average change in BMWP difference before and after interventions.

Catchment Management

The Tarland Catchment Initiative has actively worked to promote management intervention at an ecologically relevant scale involving whole tributaries rather than short stream sections (e.g., Walsh and Kunapo, 2009; Palmer et al., 2010). This is in contrast to current mechanisms of reducing diffuse pollution through management at individual land holdings and farms. Where these isolated management units do not extend to whole tributaries, it may be difficult to achieve the desired catchment-wide improvements in water quality or ecology.

Added benefits to the intervention and the potential for including more wide-ranging, multi-issue benefits have been brought about by the recognition, involvement, and participation of a range of stakeholders. The visual establishment of the buffer strips and new habitats (wetlands) as well as demonstrable success in point source clean up (e.g., the local waste water treatment plant; Stutter et al., [2010]) have resulted in an increased willingness to expand the measures to other tributaries in the catchment. The provision of objective data (e.g., Stutter et al., 2007, 2010) through which the changes can be quantified and shown is an important element in discussions of future developments with the stakeholders. The general lack of rapid positive outcomes regarding BMPs (Meals et al., 2010), here installation of buffer strips (or our inability

to detect improvement for methodological reasons), may not help the relatively slow pace of implementation and supply of high-resource requirements necessary to ensure adequate consultation and participation. A lack of rapid measurable success of the intervention could jeopardize stakeholders' interest and involvement in the work and willingness to continue with the work and to extend it across the catchment.

Although riparian fencing may improve stream water quality, it may also result in a decrease of plant and animal diversity in nutrient-rich soils (Alexander et al., 2010). Fencing is also expensive, and alternative solutions should be sought, such as reducing fertilizer input and cattle density (Alexander et al., 2010), providing off-stream water sources (Sheffield et al., 1997), and better management of critical source areas within fields (Lucci et al., 2010). This is important because the main barriers to uptake of agri-environmental schemes is mitigation costs to farmers (Bewsell et al., 2007; Lankester et al., 2009), land availability (Alexander and Allan, 2007), and ease of implementation (Gruar et al., 2010). These aspects should strongly motivate catchment management cost-effectiveness studies (e.g., Bryan and Kandulu, 2009). However, money is not the only issue; aesthetics, social, and ethical issues matter as well (Kenwick et al., 2009; Lankester et al., 2009; Spash et al., 2009).

Conclusions

Despite the apparent strengths of this study (medium term, large spatial scale), there are uncertainties regarding rapid improvements in chemical and ecological metrics as a result of widespread implementation of buffer strips. How long do we need to wait to see macroinvertebrate indices improve? How much of the catchment do we need to restore to achieve our targets? What are the costs and benefits associated with this large-scale restoration program? Is it fit for purpose? How will the behavior of the stakeholders change after these results are presented to them? Do we need to change strategy, evolving as we learn from other experiments?

Although the participative approach advocated by the WFD may prevail in the future, our study shows that robust natural science and effective communication are the pillars on which consensus may be sought regarding economic, social, and ethical issues.

Acknowledgments

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References

Alexander, G.G., and J.D. Allan. 2007. Ecological success in stream restoration: Case studies from the midwestern United States. *Environ. Manage.* 40:245–255. doi:10.1007/s00267-006-0064-6

Alexander, K., G. Foster, and N. Sanderson. 2010. "Good ecological status" of inland waterbodies: fencing of riverbanks is not "good for biodiversity." *British Wildlife* 21:326–332.

Armitage, P.D., D. Moss, J.F. Wright, and M.T. Furse. 1983. The performance of a new biological water quality score system based on macroinvertebrates over a wide range of unpolluted running-waters. *Water Res.* 17:333–347. doi:10.1016/0043-1354(83)90188-4

Arthington, A.H., R.J. Naiman, M.E. McClain, and C. Nilsson. 2010. Preserving the biodiversity and ecological services of rivers: New challenges and research opportunities. *Freshwater Biol.* 55:1–16. doi:10.1111/j.1365-2427.2009.02340.x

Benstead, J.P., A.C. Green, L.A. Deegan, B.J. Peterson, K. Slavik, W.B. Bowden, and A.E. Hershey. 2007. Recovery of three arctic stream reaches from experimental nutrient enrichment. *Freshwater Biol.* 52:1077–1089. doi:10.1111/j.1365-2427.2007.01723.x

Bernhardt, E.S., M.A. Palmer, J.D. Allan, G. Alexander, K. Barnas, S. Brooks, J. Carr, S. Clayton, C. Dahm, J. Follstad-Shah, D. Galat, S. Gloss, P. Goodwin, D. Hart, B. Hassett, R. Jenkinson, S. Katz, G.M. Kondolf, P.S. Lake, R. Lave, J.L. Meyer, T.K. O'Donnell, L. Pagano, B. Powell, and E. Sudduth. 2005. Ecology: Synthesizing US river restoration efforts. *Science* 308:636–637. doi:10.1126/science.1109769

Bewsell, D., R.M. Monaghan, and G. Kaine. 2007. Adoption of stream fencing among dairy farmers in four New Zealand catchments. *Environ. Manage.* 40:201–209. doi:10.1007/s00267-006-0184-z

Bishop, P.L., W.D. Hively, J.R. Stedinger, M.R. Rafferty, J.L. Lojpersberger, and J.A. Bloomfield. 2005. Multivariate analysis of paired watershed data to evaluate agricultural best management practice effects on stream water phosphorus. *J. Environ. Qual.* 34:1087–1101. doi:10.2134/jeq2004.0194

Bryan, B.A., and J.M. Kandulu. 2009. Cost-effective alternatives for mitigating *Cryptosporidium* risk in drinking water and enhancing ecosystem services. *Water Resour. Res.* 45:W08437. doi:10.1029/2008WR007606

Carpenter, S.R., T.M. Frost, D. Heisey, and T.K. Kratz. 1989. Randomized intervention analysis and the interpretation of whole-ecosystem experiments. *Ecology* 70:1142–1152. doi:10.2307/1941382

Clews, E., and S.J. Ormerod. 2010. Appraising riparian management effects on benthic macroinvertebrates in the Wye River system. *Aquat. Conserv.* 20:S73–S81. doi:10.1002/aqc.1094

Cooper, R.J., R.C. Ferrier, R.D. Harmel, S.J. Langan, A.J.A. Vinten, and M.I. Stutter. 2006. An initial assessment of the suitability of total maximum daily loads (TMDLs) as a means of managing diffuse pollution under the water framework directive. p. 45–52. *In* L. Garins, K. Crighton and W. Jeffrey (ed.) Agriculture and the environment VI: Managing diffuse agricultural pollution. Proc. of the SAC and SEPA Biennial Conf., Edinburgh, UK.

Cuttle, S.P., C.J.A. Macleod, D.R. Chadwick, D. Scholefield, P.M. Haygarth, P. Newell-Price, D. Harris, and M.A. Shepherd. 2007. An inventory of methods to control diffuse water pollution from agriculture (DWPA). User manual. Prepared as part of Defra project ES0203, UK. Available at http://www.swarmhub.co.uk/downloads/pdf/Diffuse_water_pollution_user_manual.pdf (verified 1 July 2011).

Davies-Colley, R.J., J.W. Nagels, R.A. Smith, R.G. Young, and C.J. Phillips. 2004. Water quality impact of a dairy cow herd crossing a stream. *N. Z. J. Mar. Fresh. Res.* 38:569–576. doi:10.1080/00288330.2004.9517262

Death, R.G., and K.J. Collier. 2010. Measuring stream macroinvertebrate responses to gradients of vegetation cover: When is enough enough? *Freshwater Biol.* 55:1447–1464. doi:10.1111/j.1365-2427.2009.02233.x

Environment Agency. 1999. Procedure for collecting and analysing macroinvertebrate samples BT001 (version 2.0). Environment Agency, Bristol, UK.

Friberg, N. 2010a. Pressure-response relationships in stream ecology: Introduction and synthesis. *Freshwater Biol.* 55:1367–1381. doi:10.1111/j.1365-2427.2010.02442.x

Friberg, N. 2010b. Ecological consequences of river channel management. p. 76–105. *In* R.C. Ferrier and A. Jenkins (ed.) Handbook of catchment management. Blackwell, Chichester, UK.

Friberg, N., J. Skriver, S.E. Larsen, M.L. Pedersen, and A. Buffagni. 2010. Stream macroinvertebrate occurrence along gradients in organic pollution and eutrophication. *Freshwater Biol.* 55:1405–1419. doi:10.1111/j.1365-2427.2008.02164.x

Gruar, D.J., A.J. Morris, C.M. Bailey, and I.A. Dillon. 2010. Development of an agri-environmental option through research trials: Skylark plots at Hope Farm. *Aspects Appl. Biol.* 100:341–346.

Harper, D.M., M. Ebrahimnezhad, E. Taylor, S. Dickinson, O. Decamp, G. Verniers, and T. Balbi. 1999. A catchment-scale approach to the physical restoration of lowland UK rivers. *Aquat. Conserv.* 9:141–157. doi:10.1002/(SICI)1099-0755(199901/02)9:1<141::AID-AQC328>3.0.CO;2-C

Hawkes, H.A. 1998. Origin and development of the Biological Monitoring Working Party (BMWP) score system. *Water Res.* 32:964–968. doi:10.1016/S0043-1354(97)00275-3

Hilton, J. 2002. A dictionary of BMPs for the reduction of silt, phosphorus

- and other pollutants emanating from diffuse sources into surface waters. National Environmental Research Council report v1.0. Centre of Ecology and Hydrology, Dorset, UK.
- Hurlbert, S.H. 1984. Pseudoreplication and the design of ecological field experiments. *Ecol. Monogr.* 54:187–211. doi:10.2307/1942661
- Kay, P., A.C. Edwards, and M. Foulger. 2009. A review of the efficacy of contemporary agricultural stewardship measures for ameliorating water pollution problems of key concern to the UK water industry. *Agric. Syst.* 99:67–75. doi:10.1016/j.agry.2008.10.006
- Kenwick, R.A., R. Shammin, and W.C. Sullivan. 2009. Preferences for riparian buffers. *Landscape Urban Plan.* 91:88–96. doi:10.1016/j.landurbplan.2008.12.005
- Langan, S.J., A.J. Wade, R. Smart, A.C. Edwards, C. Soulsby, M.F. Billett, H.P. Jarvie, M.S. Cresser, R. Owen, and R.C. Ferrier. 1997. The prediction and management of water quality in a relatively unpolluted catchment: Current issues and experimental approaches. *Sci. Total Environ.* 194–195:419–435. doi:10.1016/S0048-9697(96)05380-6
- Lankester, A., R. Valentine, and A. Cottrell. 2009. 'The sweeter country': Social dimensions to riparian management in the Burdekin rangelands, Queensland. *Australas. J. Environ. Manage.* 16:94–102.
- Larsen, S., and S.J. Ormerod. 2010a. Low-level effects of inert sediments on temperate stream invertebrates. *Freshwater Biol.* 55:476–486. doi:10.1111/j.1365-2427.2009.02282.x
- Larsen, S., and S.J. Ormerod. 2010b. Combined effects of habitat modification on trait composition and species nestedness in river invertebrates. *Biol. Conserv.* 143:2638–2646. doi:10.1016/j.biocon.2010.07.006
- Larsen, S., G. Pace, and S.J. Ormerod. 2011. Experimental effects of sediment deposition on the structure and function of macroinvertebrate assemblages in temperate streams. *River Res. Appl.* 27:257–267. doi:10.1002/rra.1361
- Larsen, S., I.P. Vaughan, and S.J. Ormerod. 2009. Scale-dependent effects of fine sediments on temperate headwater invertebrates. *Freshwater Biol.* 54:203–219. doi:10.1111/j.1365-2427.2008.02093.x
- Lucci, G.M., R.W. McDowell, and L.M. Condron. 2010. Potential phosphorus and sediment loads from sources within a dairy farmed catchment. *Soil Use Manage.* 26:44–52. doi:10.1111/j.1475-2743.2009.00247.x
- Malakoff, D. 2004. Profile: Dave Rosgen-The river doctor. *Science* 305:937–939. doi:10.1126/science.305.5686.937
- Matthaei, C.D., J.J. Piggott, and C.R. Townsend. 2010. Multiple stressors in agricultural streams: Interactions among sediment addition, nutrient enrichment and water abstraction. *J. Appl. Ecol.* 47:639–649. doi:10.1111/j.1365-2664.2010.01809.x
- Meals, D.W., S.A. Dressing, and T.E. Davenport. 2010. Lag time in water quality response to best management practices: A review. *J. Environ. Qual.* 39:85–96. doi:10.2134/jeq2009.0108
- Miller, J., D. Chanasyk, T. Curtis, T. Entz, and W. Willms. 2010a. Influence of streambank fencing with a cattle crossing on riparian health and water quality of the Lower Little Bow River in Southern Alberta, Canada. *Agric. Water Manage.* 97:247–258. doi:10.1016/j.agwat.2009.09.016
- Miller, J.J., D.S. Chanasyk, T. Curtis, and W.D. Willms. 2010b. Influence of streambank fencing on the environmental quality of cattle excluded pastures. *J. Environ. Qual.* 39:991–1000. doi:10.2134/jeq2009.0233
- Nakano, S., and M. Murakami. 2001. Reciprocal subsidies: Dynamic interdependence between terrestrial and aquatic food webs. *Proc. Natl. Acad. Sci. USA* 98:166–170. doi:10.1073/pnas.98.1.166
- Ormerod, S.J. 2004. A golden age of river restoration science? *Aquat. Conserv.* 14:543–549. doi:10.1002/aqc.663
- Ormerod, S.J., and I. Durance. 2009. Restoration and recovery from acidification in upland Welsh streams over 25 years. *J. Appl. Ecol.* 46:164–174. doi:10.1111/j.1365-2664.2008.01587.x
- Owens, L.B., W.M. Edwards, and R.W. VanKeuren. 1996. Sediment losses from a pastured watershed before and after stream fencing. *J. Soil Water Conserv.* 51:90–94.
- Palmer, M.A., E.S. Bernhardt, J.D. Allan, P.S. Lake, G. Alexander, S. Brooks, J. Carr, S. Clayton, C.N. Dahm, J.F. Shah, D.L. Galat, S.G. Loss, P. Goodwin, D.D. Hart, B. Hassett, R. Jenkinson, G.M. Kondolf, R. Lave, J.L. Meyer, T.K. O'Donnell, L. Pagano, and E. Sudduth. 2005. Standards for ecologically successful river restoration. *J. Appl. Ecol.* 42:208–217. doi:10.1111/j.1365-2664.2005.01004.x
- Palmer, M.A., H.L. Menninger, and E. Bernhardt. 2010. River restoration, habitat heterogeneity and biodiversity: A failure of theory or practice? *Freshwater Biol.* 55:205–222. doi:10.1111/j.1365-2427.2009.02372.x
- Parkyn, S.M., R.J. Davies-Colley, N.J. Halliday, K.J. Costley, and G.F. Crocker. 2003. Planted riparian buffer zones in New Zealand: Do they live up to expectations? *Restor. Ecol.* 11:436–447. doi:10.1046/j.1526-100X.2003.rec0260.x
- Ranganath, S.C., W.C. Hession, and T.M. Wynn. 2009. Livestock exclusion influences on riparian vegetation, channel morphology, and benthic macroinvertebrate assemblages. *J. Soil Water Conserv.* 64:33–42. doi:10.2489/jswc.64.1.33
- Schalchli, U. 1992. The clogging of coarse gravel beds by fine sediment. *Hydrobiologia* 235–236:189–197. doi:10.1007/BF00026211
- SEERAD. 2005. Prevention of environmental pollution from agricultural activity: A code of good practice. Scottish Executive, Edinburgh, UK.
- SEPA. 2000. The Dee Catchment Management Plan issues consultation document. Scottish Environmental Protection Agency, Aberdeen, Scotland.
- Sheffield, R.E., S. Mostaghimi, D.H. Vaughan, E.R. Collins, and V.G. Allen. 1997. Off-stream water sources for grazing cattle as a stream bank stabilization and water quality BMP. *Trans. ASAE* 40:595–604.
- Spash, C.L., K. Urama, R. Burton, W. Kenyon, P. Shannon, and G. Hill. 2009. Motives behind willingness to pay for improving biodiversity in a water ecosystem: Economics, ethics, and social psychology. *Ecol. Econ.* 68:955–964. doi:10.1016/j.ecolecon.2006.09.013
- Stutter, M.I., B.O.L. Demars, and S.J. Langan. 2010. River phosphorus cycling: Separating biotic and abiotic uptake during short-term changes in sewage effluent loading. *Water Res.* 44:4425–4436. doi:10.1016/j.watres.2010.06.014
- Stutter, M.I., S.J. Langan, and B.O.L. Demars. 2007. River sediments provide a link between catchment pressures and ecological status in a mixed land use Scottish River system. *Water Res.* 41:2803–2815. doi:10.1016/j.watres.2007.03.006
- Stutter, M.I., S.J. Langan, and R.J. Cooper. 2008. Spatial and temporal dynamics of stream water particulate and dissolved N, P and C forms along a catchment transect, NE Scotland. *J. Hydrol.* 350:187–202. doi:10.1016/j.jhydrol.2007.10.048
- Stutter, M.I., S.J. Langan, and D.G. Lumsdon. 2009. Vegetated buffer strips can lead to increased release of phosphorus to waters: A biogeochemical assessment of the mechanisms. *Environ. Sci. Technol.* 43:1858–1863. doi:10.1021/es8030193
- Sweeney, B.W., T.L. Bott, J.K. Jackson, L.A. Kaplan, J.D. Newbold, L.J. Standley, W.C. Hession, and R.J. Horwitz. 2004. Riparian deforestation, stream narrowing, and loss of stream ecosystem services. *Proc. Natl. Acad. Sci. USA* 101:14132–14137. doi:10.1073/pnas.0405895101
- ter Braak, C.J.F., and P. Smilauer. 2002. *Canoco 4.5 reference manual and CanoDraw for Windows. User's guide to Canoco for Windows: Software for canonical community ordination (vers. 4.5).* Microcomputer Power, Ithaca, NY.
- Townsend, C.R., S.S. Uhlmann, and C.D. Matthaei. 2008. Individual and combined responses of stream ecosystems to multiple stressors. *J. Appl. Ecol.* 45:1810–1819. doi:10.1111/j.1365-2664.2008.01548.x
- UK TAG. 2008. UK Environmental Standards and Conditions (Phase 1). UK Technical Advisory Group, Final Report. Available at: http://www.wfduk.org/UK_Environmental_Standards/LibraryPublicDocs/UKTAG%20ReportAug%202006UKEnvironmentalStandardsandConditionsFinalReport (verified 1 July 2011).
- USEPA. 2000. National water quality inventory. Office of Water, USEPA, Washington, DC.
- Vinten, A.J.A., M.I. Stutter, S. Dunn, J. Potts, J. MacDonald, F. Napier, W. Jeffrey, and C. Birkel. 2010. How effective is the implementation of controls on diffuse pollution under the Water Framework Directive in Scotland? Answers and questions from the Lunan water diffuse pollution monitored catchment project. p. 80–83. *In Proc. of the Water Quality 2010 Conf.*, Leeds University, Leeds, UK, 23–24 June 2010.
- Walsh, C.J., and J. Kunapo. 2009. The importance of upland flow paths in determining urban effects on stream ecosystems. *J. North Am. Benthol. Soc.* 28:977–990. doi:10.1899/08-161.1
- Wood, P.J., and P.D. Armitage. 1997. Biological effects of fine sediment in the lotic environment. *Environ. Manage.* 21:203–217. doi:10.1007/s002679900019
- Wright, J.F., M.T. Furse, P.D. Armitage, and D. Moss. 1993. New procedures for identifying running water sites subject to environmental stress and for evaluating sites for conservation, based on the macroinvertebrate fauna. *Arch. Hydrobiol.* 127:319–326.
- Wright, J.F., D. Moss, P.D. Armitage, and M.T. Furse. 1984. A preliminary classification of running water sites in Great Britain based on macro-invertebrate species and prediction of community types using environmental data. *Freshwater Biol.* 14:221–256. doi:10.1111/j.1365-2427.1984.tb00039.x
- Wright, J.F., D.W. Sutcliffe, and M.T. Furse. 2000. Assessing the biological quality of fresh waters: RIVPACS and other techniques. *Freshwater Biological Association, Cumbria, UK.*
- Zalewski, M., D.M. Harper, B.O.L. Demars, G. Jolánkai, G. Crosa, G.A. Janauer, and N. Pacini. 2008. Linking biological and physical processes at the river basin scale: The origins, scientific background and scope of ecohydrology. p. 1–17. *In D.M. Harper, M. Zalewski, and N. Pacini (ed.) Ecohydrology: Processes, models and case studies.* CAB Int., Wallingford, UK.