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Biological assessment of recreation-associated impacts on the water quality of streams crossing the West Highland Way, Scotland

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ABSTRACT
A study was carried out in summer 2012 to assess the potential scale of recreation-associated impact upon streams crossing or adjacent to the West Highland Way, Scotland, using benthic macroinvertebrates as an indicator of water quality. Differences in water quality between sites located downstream and upstream of the footpath were considered for 22 streams. The results showed the presence of at least four recognisably different macroinvertebrate communities in these streams, indicating differing standards of water quality, from moderately good to poor, but provided little or no evidence of human impact from recreational activities (including wild camping) associated with the West Highland Way. Rather, the results suggest that differences at stream catchment scale, most likely related to natural factors (e.g. differences in soils, geology and relief) and catchment land-use, are more likely to be the cause of the observed differences in invertebrate communities and bioassessed water quality.

INTRODUCTION

In 1980, the West Highland Way (WHW) was opened in Scotland as a long distance walking route (154 km in length) between Glasgow and Fort William (den Breejen, 2007). There are many streams and rivers which cross or run adjacent to the Way (http://www.west-highland-way.co.uk). With an annually-estimated 67,000 people either completing the WHW, or using parts of it for shorter walks (den Breejen, 2007), these watercourses (which in common with most upland streams in Scotland would be expected to be of good quality: Gilvear et al., 2002) have the potential to suffer local pollution. This is particularly the case where wild campsites are set up by walkers, and streams are used for water supply and washing purposes, resulting in direct and indirect pollution of the freshwater environment through the transfer of substances such as sun creams, soap and detergent, insect repellent, food particles, litter, and faecal waste, from both humans and accompanying animals, mainly dogs (Derlet et al., 2008).

Concern over this issue, along the section of the WHW located in the Loch Lomond & The Trossachs National Park led the Loch Lomond & The Trossachs National Park Authority (2012a, b) to act to try to reduce the problems caused by wild camping (http://www.lochlomond-trossachs.org), with the introduction of the 'East Loch Lomond Camping Byelaws' which make it illegal for anyone to camp within the East Loch Lomond Restricted Zone at any time between 1st March and 31st October unless they are camping within an official designated camp site. Further north wild camping is permitted along the WHW and there are also some streamside "designated free wild campsite" sites (without facilities), for example at Inveroran and Kingshouse (see Table 1).

To assess the potential local impacts of recreation-associated activities, particularly walking and associated wild camping, upon the ecology of streams crossing, or adjacent to, the WHW, a study was undertaken, during summer 2012, of streams along the length of the path. The survey utilised benthic macroinvertebrates as an indicator of water quality. Macroinvertebrates are commonly used for this purpose in freshwater systems worldwide (e.g. Chessman, 1995; Smith et al., 1999; Brown, 2001; Nicholas & Norris, 2006), and a large number of individual metrics have been developed for this purpose, all based upon differences in the sensitivity of benthic macroinvertebrate families to water pollution.. Of these bioindicator protocols one of the most widely-used is the BMWP approach (Biological Monitoring Working Party score system: Hawkes, 1998), and this was adopted for the purposes of this study. Differences in water quality immediately downstream and upstream of the long-distance footpath were assessed at a range of sites, in streams with varying catchment land-use, along the full length of the WHW.

MATERIALS & METHODS

Sampling was carried out between 21/05/2012 and 15/06/2012. Ordnance Survey 1:50,000 maps were used to locate streams crossing or adjacent to the route of the WHW. In total 22 sample sites were selected, based on their location, ease of access, catchment land-use, and whether there was evidence that camping had taken place adjacent to the stream (Table 1; Fig. 1). Evidence included direct observation of tents, or of typical camping disturbance to vegetation, presence of fire-sites, camping-associated litter (food wrappers, bottles, cans, toilet paper etc.), as well as the presence of signposted "designated free wild campsite" sites, for example at Bridge of Orchy and Inveroran.

Table 1. Sampling site information: NGR: UK National Grid Reference; d/s downstream; u/s upstream. Catchment land-use: A = agriculture; M = moorland; F = forestry; U = urban or semi-urban.

Site	Name	Sample	Sample code number	Stream name	NGR	Catchment land-use	Camping evidence?
1	Dumgoyach	1A d/s 1B u/s	1 2	Blane Water	NS530815	A, M, F, U	none
2	Garadhban Forest	2A d/s 2B u/s	3 4	unnamed stream	NS475908	F, M	none
3	Breac Leac	3A d/s 3B u/s	5 6	Burn of Mar	NS445925	M	unofficial wild camping
4	Millarochy Car Park	4Ad/s 4B u/s	7 8	unnamed stream	NS411922	F, M	none
5	Cashel	5A d/s 5B u/s	9 10	Cashell Burn	NS395541	F, M	adjacent to commercial campsite
6	Sallochy Bay Car Park	6A d/s 6B u/s	11 12	unnamed stream	NS380958	F	none
7	Lochan Maol Dhuinne	7A d/s 7B u/s	13 14	Caol Ghleann Stream	NS367971	F, M	unofficial wild camping
8	Rowardennan Lodge	8A d/s 8B u/s	15 16	unnamed stream	NS359992	F, M	none
9	Cailness	9A d/s 9B u/s	17 18	unnamed stream	NN342062	F, M	unofficial wild camping
10	Inversnaid (north)	10A d/s 10B u/s	19 20	unnamed stream	NN336095	F, M	none
11	Doune (south)	11A d/s 11B u/s	21 22	unnamed stream	NN333136	F, M	unofficial wild camping
12	Ardleish	12A d/s 12B u/s	23 24	unnamed stream	NN328158	M	none
13	Beinglas	13A d/s 13B u/s	25 26	Ben Glas Burn	NN321187	M	adjacent to commercial campsite
14	Kirkton	14A d/s 14B u/s	27 28	unnamed stream	NN359282	A, M	none
15	Auchtertyre	15A d/s 15B u/s	29 30	Allt Gleann a'Chlachain	NN353290	M	adjacent to commercial campsite
16	Dalrigh	16A d/s 16B u/s	31 32	River Fillan	NN345288	A, F, M	none
17	Tyndrum	17A d/s 17B u/s	33 34	unnamed stream	NN327303	F, M	none
18	Bridge of Orchy	18A d/s 18B u/s	35 36	River Orchy	NN296398	F, M	adjacent to designated free wild campsite
19	Inveroran	19A d/s 19B u/s	37 38	Allt Tolaghan	NN272416	A, M	adjacent to designated free wild campsite
20	Blackrock	20A d/s 20B u/s	39 40	unnamed stream	NN268536	M	none
21	Kingshouse	21A d/s 21B u/s	41 42	River Etive	NN261548	M	adjacent to designated free wild campsite
22	Allt a Lairige Moire	22A d/s 23B u/s	43 44	unnamed stream	NN099659	M	none

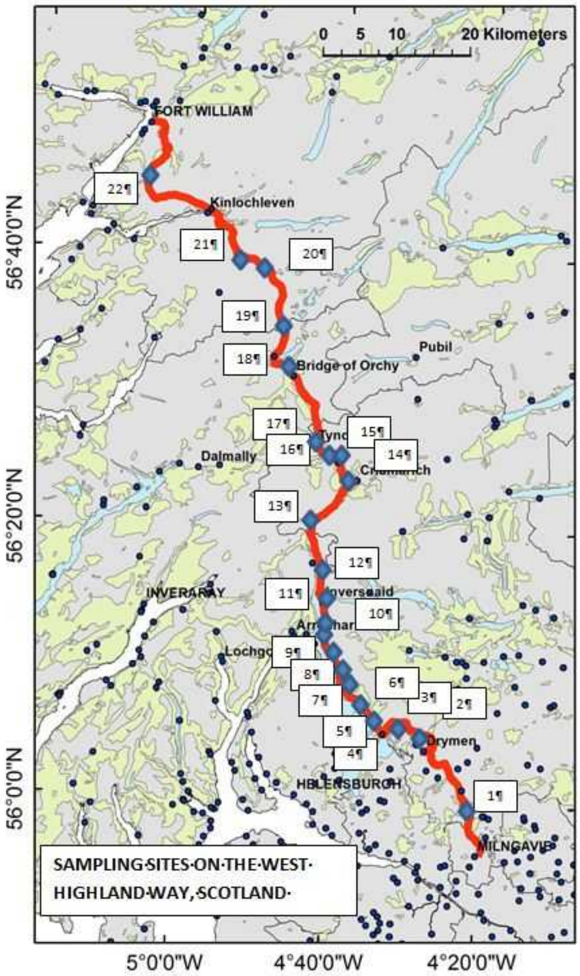

Fig. 1. Location of sampling sites along the West Highland Way. For site grid references see Table 1.

At each site, kick sampling (see Fig. 2), a standard approach for use in benthic invertebrate river bioassessment protocols (e.g. Moore & Murphy, 2015) was undertaken to collect benthic invertebrates, and environmental variables were measured, at two sub-sites, upstream and downstream of where the WHW crossed the stream, or within 10 m of the path, at sites adjacent to the WHW. The sampling protocol followed the European Standards (CEN) recommendations "Water quality – guidance on pro-rata multi-habitat sampling of benthic invertebrates from wadeable rivers: EN 16150" (British Standards Institute, 2012). Where there was evidence of camping taking place, the downstream sample was located downstream of both the camping area and the footpath crossing point (bridge or ford), and the upstream sample likewise upstream of both. Downstream sites were always sampled first at each site. In total 44 invertebrate samples were taken. Initially the net (1 mm mesh size) was dragged across the water surface of the sample area for 30 seconds. This was done in order to collect any surface dwelling organisms (e.g. Gerridae). The organisms were transferred to a sample pot where they were preserved in ethanol until analysis. A 3-minute kick sample was carried out, moving diagonally across the water in an upstream direction. The contents of the net were transferred to the same sample pot as before. In the laboratory samples were sorted through, removing all of the animals present, then organisms were identified to family level (in accordance with standard protocol for the use of benthic invertebrates for water quality assessment: e.g. Barbour et al., 1999: Metzeling et al., 2003), using appropriate identication guides (Quigley, 1977; .Pawley et al., 2011), and their total numbers were recorded.

Fig. 2. Kick sampling for benthic invertebrates: Ben Glas Burn, Site 13B, June 2012.

For each sampling site, a standard index of water quality (using BMWP scores for each family encountered: Hawkes, 1998; Centre for Intelligent Environmental Systems, 2004) was calculated from the invertebrate data. A higher BMWP score indicates higher water quality. The normal interpretation of BMWP scores suggests that a score in the range > 70 indicates good quality; 41 – 70 shows moderate water quality; 11 – 40, poor quality; and <11 polluted water (Hawkes, 1998 Clarke, et al., 2002; Sandin & Hering, 2004).

Ten environmental variables were recorded at each site. Conductivity (Cond: µS cm^{-1}) and pH were measured in situ using field meters. The average depth (AD m) of the section of stream kick-sampled was determined by 10 random measurements using a meter stick, and stream width (Wid m) was measured using a surveyor's tape. Shade (SH) from bankside vegetation, steep banks and/or bridges was assessed visually on a 3-point scale (1 = no shade; 2 = moderate shade; 3 = heavy shade: e.g. closed overhead tree canopy). Flow (F) was also assessed subjectively on a three point scale (1 = slow, "pool"; 2 = moderate flow, "glide"; 3 = fast flow, "riffle, or white-water visible"). The substrate composition was visually estimated, recording the approximate percentage cover of boulders/bedrock and cobbles (%SCbc), pebbles and gravel (%SCpg), sand (%SCsa), and silt (% SCsi).

The macroinvertebrate families present, and environmental data, were analysed using two multivariate programs to classify and ordinate the data. Canonical Correspondence Analysis (CCA: ter Braak & Šmilauer, 1998) was carried out with the macroinvertebrate family abundance data constrained by the environmental data as a way of evaluating the relationships between samples, family and environmental variables. This could only be carried out using those environmental variables for which there was a full dataset, and since pH and conductivity were not measured at some sites (due to equipment malfunction), only eight environmental variables were included in the CCA ordination. Monte Carlo testing was used to assess the significance of correlations identified by CCA between environmental variables and macroinvertebrate families, and also between the environmental variables and samples, across all canonical axes of the ordination.

Two Way Indicator Species Analysis (TWINSPAN: Hill, 1979) was used to classify the samples based on the presence of different macroinvertebrate families. Samples that supported similar invertebrate communities in terms of family composition were split into discrete end-groups by the analysis. TWINSPAN also identified those families which characterise ("indicate") each sample end-group.,
Ryan-Joiner testing was used to assess normality for each environmental dataset, and square root, natural log, or $\log_{10}$ transformations were applied, as appropriate, to normalise the data where necessary. Analysis of Variance (ANOVA), with subsequent separation of means, for significant ($P<0.05$) ANOVA outcomes, using Tukey's mean comparison test, was used in order to test for significance of means of the individual variables (environmental variables and BMWP score), between sets of samples making up the TWINSPAN end-groups. Paired t-tests were used to compare downstream v. upstream sample BMWP scores. Variables that could not be normalised underwent Kruskal-Wallis non-parametric testing, to test for significance of medians of each variable between the sample-groups.

RESULTS

In total 26 benthic invertebrate families were recorded at the sample sites (see caption to Fig. 3 for list of families).The CCA outcome showed the relationships between family occurrence and eight environmental variables (Fig. 3), whilst the result of Monte Carlo testing, for all canonical axes of the ordination (P = 0.004), indicated that the variation explained by the CCA results was significant, across all ordination axes combined.

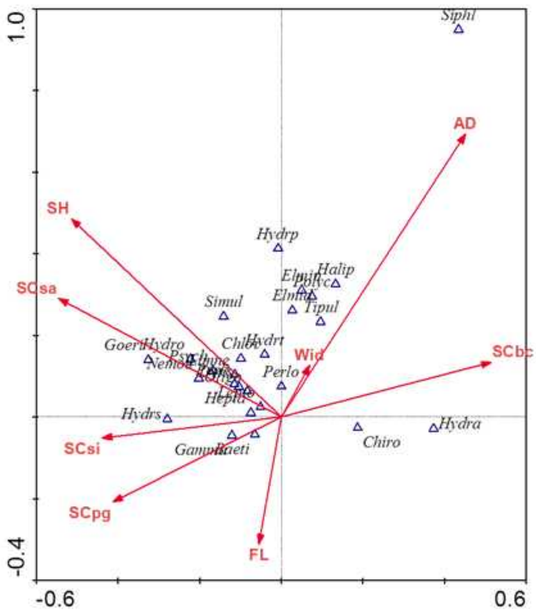

Fig. 3. CCA ordination plot for macroinvertebrate family-environmental analysis (26 macroinvertebrate taxa collected from streams and rivers on the West Highland Way). Monte Carlo test outcome, axis 1 (horizontal axis): P=0.002; all canonical axes: P=0.004. Eigenvalues: axis 1 (horizontal): 0.304; axis 2 (vertical): 0.160. Environmental variable codes: Wid = Width, AD = Average Depth, FL = Flow, SH = Shade, SCb c= Substrate Composition: boulders & cobbles, SCpg = Substrate Composition: pebbles & gravel, SCsa = Substrate Composition: Sand, SCsi = Substrate Composition: Silt. Taxa codes: Baeti= Baetidae, Chiro = Chironomidae, Chloro = Chloroperlidae, Dytis = Dytiscidae, Elmid = Elmidae, Elmin = Elminthidae, Gamma = Gammaridae, Goeri = Goeridae, Halip = Haliplidae, Hepta = Heptageniidae, Hydra = Hydraenidae, Hydro = Hygrobiidae, Hydrp = Hydrophilidae, Hydrs = Hydropsychidae, Hydrt = Hydroptilidae, Lepto = Leptophlebiidae, Limne = Limnephilidae, Nemou = Nemouridae, Oligo = Oligochaeta, Perli = Perlidae, Perlo = Perlodidae, Polyc = Polycentropidae, Psych= Psychomyiidae, Simul = Simulidae, Siphl = Siphlonuridae, Tipul = Tipulidae.

Long environmental vector arrows in the CCA ordination plot are an indication of relatively greater importance of a variable in potentially driving family distribution in the sample streams, while the direction of the arrows indicates the primary gradient of a particular variable through the ordination space. The position of families within the plot, relative to the environmental gradients, provides evidence of likely associations between environmental factors and family habitat preferences. On this evidence depth (AD) was the most important environmental predictor for benthic invertebrates in the streams sampled, shade (SH) second, and width (Wid) least important. The eigenvalues for the CCA principal axes were low (see Fig. 3) suggesting that even the longest gradient was not indicative of major environmental differences between the sample sites, though the significant Monte Carlo test result indicates that variation was nevertheless non-random. Substrate composition (SC) and flow (FL) were of intermediate importance in prediction of family distribution. Some families were particularly associated with certain environmental conditions. For example, Fig. 3 shows that the mayfly family Siphlonuridae (Siphl) tended to be found in deeper waters, whilst the water beetle family Hydraenidae (Hydra) was most closely associated with hard substrates (bare rock and cobbles: SCbc).

TWINSPAN analysis identified four end sample-groups. Group I (with 17 samples) was indicated by the presence of Chironomidae, Heptageniidae, and Polycentropidae. Group II (10 samples) had Chloroperlidae and Simulidae as indicators. Group III (14 samples) was indicated by Baetidae, Heptageniidae, and Gammaridae, while the small Group IV (only 3 samples) had no indicator families (see caption to Fig. 4 for sample membership of groups). All sample-groups contained a mix of both downstream and upstream samples from the survey sites. There is considerable overlap between TWINSPAN groups on the ordination diagram, which reflects the relatively low eigenvalues recorded for the TWINSPAN classification divisions producing the 4 end-groups (eigenvalue range: 0.313 – 0.340). Low eigenvalues indicate a high degree of family overlap between the samples comprising the end-groups. However there is a tendency for Group I samples to occupy preferentially the right-hand side of the ordination, associated with higher flow and coarser substrate particle size); whilst Group II and III samples tend to occur more towards the left (typified by relatively deeper water and finer substrates). Group IV samples occupy an intermediate position on the sample ordination plot.

Although weak trends were detected by the ordination for substrate size and flow, statistical analysis (ANOVA) of differences in environmental and biotic variables between TWINSPAN sample-groups showed no significant differences in flow or substrate composition across the four sample-groups. However, there were between-group differences for the remaining variables (Table 2, Table 3).

Considering the sample-groups in decreasing order of mean BMWP score, Group III samples had the highest mean values for both invertebrate diversity and BMWP score,

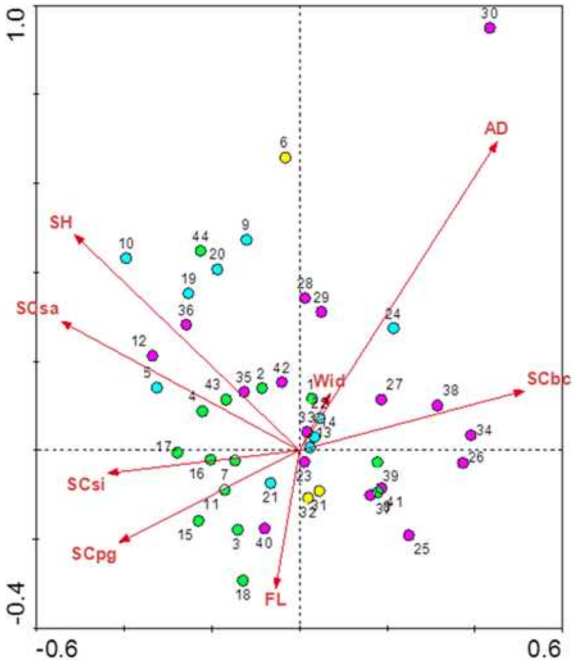

Fig. 4. CCA ordination plot for sample-environmental analysis of 44 samples collected from streams and rivers located on the West Highland Way. For ordination statistics and environmental variable codes see Figure 2. TWINSPAN sample-groups hare highlighted: Group I (purple) = samples 12, 23, 25, 26, 27, 28, 29, 30, 33, 34, 35, 36, 37, 38, 40, 41, 42; Group II (blue) = 5, 9, 10, 13, 14, 19, 20, 21, 22, 24; Group III (green) = 1, 2, 3, 4, 7, 8, 11, 15, 16, 17, 18, 39, 43, 44; Group IV (yellow) = 6, 31, 32. See Table 1 for more information on sample locations.

both suggesting moderately good water quality. This set of samples was typically from narrow, shallow streams with moderately high conductivity (comparable to values found in the South Basin of Loch Lomond: Habib et al., 1997), and circumneutral to slightly acidic pH. The samples measured in the unnamed stream (draining a wholly conifer-afforested catchment: see Table 1) flowing into Loch Lomond through Millarochy Car Park (Site 4: samples 7 and 8) were the most acidic of any of the streams surveyed at c. pH 5.0. Group III samples included the Blane Water in the south, and a scattered set of streams, throughout the length of the WHW, to the northernmost site sampled (Site 22). Only one Group I site was potentially influenced by camping activities (the downstream site at Cailness, sample 17: Table 1). With the exception of the Blane Water, these samples were all from small streams draining ,moorland or forest catchments with little or no agricultural influence (see Table 1).

Table 2. Mean values (± 1 standard error) of stream depth, conductivity, family diversity and BMWP score showing the differences between TWINSPAN sample-groups I - IV, as shown by significant one-way ANOVA outcome (*P* <0.05) and subsequent application of Tukey's mean separation test. Mean values (per environmental factor) sharing a letter in common are not significantly different from each other. Significance: * *P*<0.05; ** *P* <0.01; *** *P* <0.001.

	ANOVA comparison of TWINSPAN sample-groups				*P* -value
	I	II	III	IV	
Average Depth (m)	$0.23^{a} \pm 0.03$	$0.18^{ab} \pm 0.02$	$0.13^{b} \pm 0.02$	$0.18^{ab} \pm 0.05$	0.01**
Conductivity ($\mu S\ cm^{-1}$)	$42.4^{c} \pm 5.6$	$38.5^{bc} \pm 5.0$	$81.6^{a} \pm 19.5$	$103.2^{abc} \pm 36.2$	0.003**
Family Diversity	$5.4^{a} \pm 0.46$	$5.8^{a} \pm 0.53$	$7.2^{a} \pm 0.67$	$1.7^{b} \pm 0.33$	0.001***
BMWP Score	$30.9^{b} \pm 2.5$	$36.3^{ab} \pm 0.8$	$47.3^{a} \pm 4.4$	$6.7^{c} \pm 3.0$	0.001***

Table 3. Summary table of Kruskal-Wallis test results comparing non-normal environmental variables between TWINSPAN sample-groups. Significance: NS not significant; * *P* <0.05; ** *P* <0.01.

Kruskal-Wallis comparison of TWINSPAN sample-groups				
	Group with highest median	Group with lowest median	*P* -value (adjusted for ties)	Significance
Width (m)	I	III & IV	0.009	**
pH	I	II	0.017	*
SC: boulders & cobbles (%)	I	IV	0.139	NS
SC: pebbles & gravel (%)	IV	I	0.088	NS
SC: sand (%)	III	I	0.142	NS
SC: silt (%)	-	-	0.348	NS
Flow	II	I & IV	0.243	NS
Shade	II & III	I & IV	0.041	*

Samples forming Groups I and II had intermediate mean BMWP scores. Group II sites tended to lie on the southern half of the WHW, generally at low altitude, and were overall more lowland in nature, whilst all but one of the Group I sites were on the northern, more upland, section of the WHW. Both these groups had similar BMWP scores, not significantly different from each other, but lower than Group III samples, (though not significantly so for Group II). Family diversity showed a similar trend, with the biotic data hence overall suggesting poorer water quality than in Group III samples. The differences in invertebrate community detected by TWINSPAN (with Group I indicated by Chironomidae, Heptageniidae, and Polycentropidae, whilst Group II indicators were Chloroperlidae and Simulidae) probably reflect these geographical differences (see Figs. 5 and 6 for examples of contrasting indicator family distributions along the course of the WHW), with the adverse influences on water quality being probably derived from differing sources for the two groups. All but one of the sites used for wild camping lay in either Group I or II, though in several cases samples located upstream of the area where evidence of camping was observed were in the same sample-group as the downstream site on that stream (e.g. Group I samples 35 and 36, 37 and 38, 41 and 42; Group II samples 13 and 14, 21 and 22).

Group IV was the smallest sample-group, and had the poorest water quality, as measured by its BMWP score, as well as low family diversity. One of the sites was adjacent to an unofficial wild camping site, on the Burn of Mar, and though the sample site was positioned upstream of the WHW crossing, and visible signs of wild camping, it is still possible that pollution from camping activities and human waste might have affected this site. The other two sites were on the R. Fillan near Dalrigh, where a substantial field drain entered the river, causing noticeable water discolouration, and probably producing at least local point-source organic pollution, highly likely to influence benthic invertebrate community composition (Alvarez-Cabria et al, 2011).

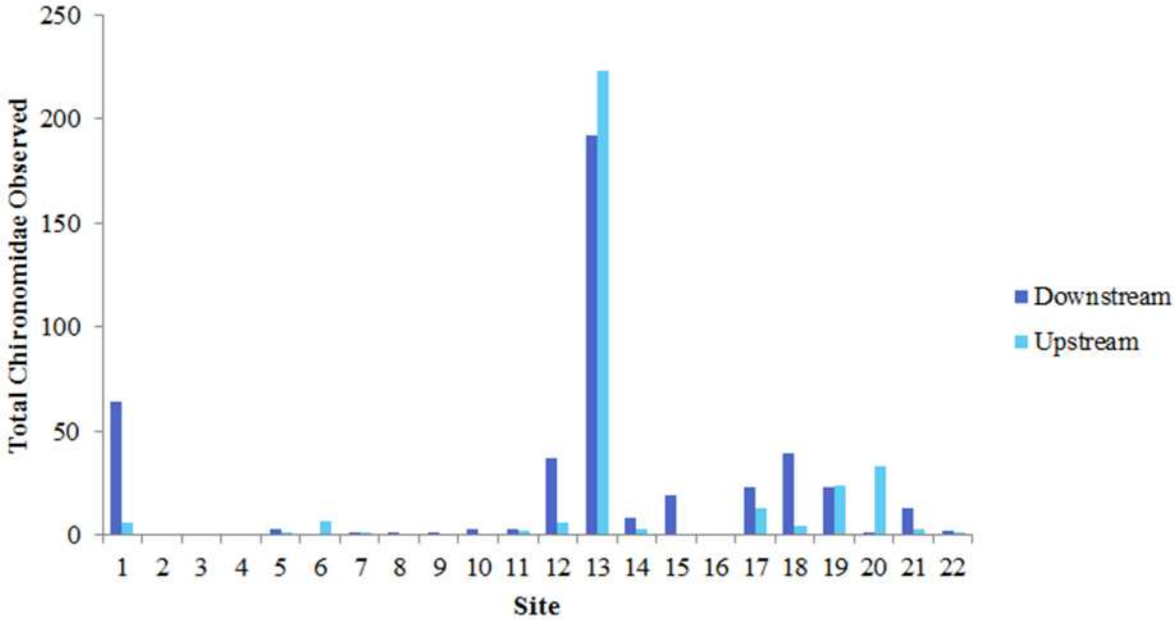

Fig. 5. Total number of Chironomidae recorded at each site. Site 1: southernmost sampling point; Site 22: northernmost.

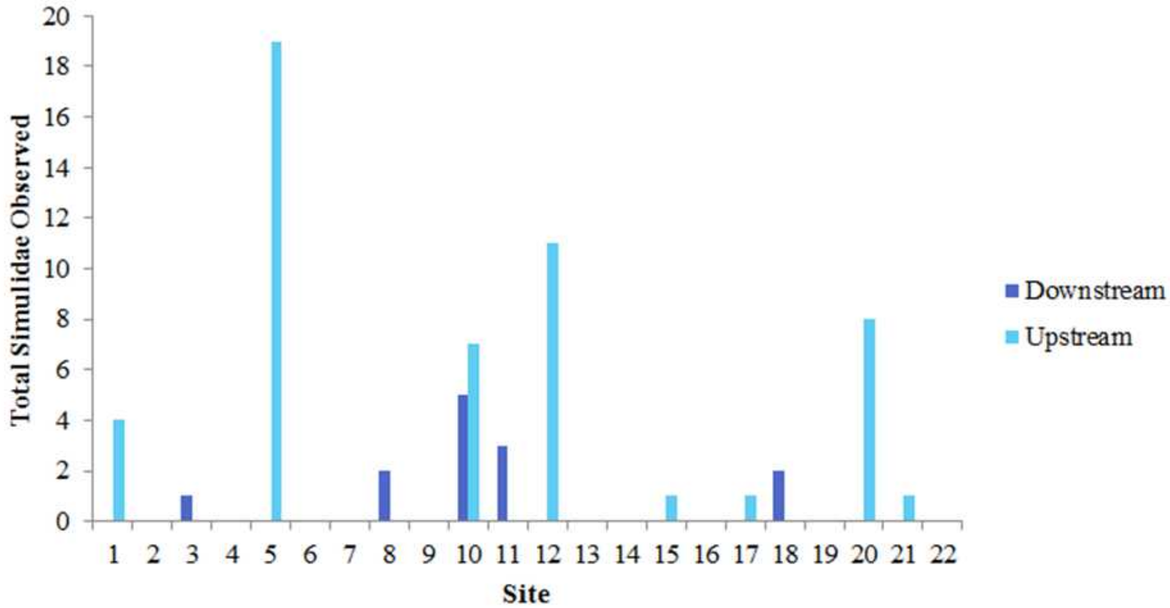

Fig. 6. Total number of Simulidae recorded at each site. Site 1: southernmost sampling point; Site 22: northernmost.

Table 4. Summary table of significant Kruskal-Wallis test results carried out using abundance data for TWINSPAN sample-group indicator families. No significant differences were found for Chloroperlodidae. Significance: NS: not significant; * P <0.05; ** P <0.01; *** P <0.001.

Family	Group with Highest Median	Group with Lowest Median	P -value (adjusted for ties)	Significance
Baetidae	I	IV	0.001	***
Chironomidae	I	IV	0.000	***
Gammaridae	III	I, II & IV	0.000	***
Heptageniidae	I	IV	0.000	***
Polycentropidae	I & III	II & IV	0.046	*
Simulidae	II	I, III & IV	0.037	*

For the families identified by TWINSPAN as sample-group indicators (Table 4), it was noticeable that amphipods (Gammaridae: Group III indicator) had significantly higher abundance at samples occurring in this group than in the other groups. The same trend was seen for the three Group I indicator families (the mayfly family, Heptageniidae; midge larvae of the Chironomidae (Fig. 5); and the caddis family Polycentropidae), all of which showed high abundance compared with other end-groups. In Group II one of the indicators, blackfly (Simulidae: Fig. 6) also showed a significant tendency towards high abundance compared with abundance in other TWINSPAN groups. Although high abundance is not necessarily a criterion for selection of indicators by TWINSPAN, in this case there seems to be a suggestion that indicator families did not just tend to occur selectively at the sites making up the end-groups of the classification, but also tended to do so at relatively high abundance.

Comparisons of BMWP scores from sites upstream and downstream of the WHW revealed no significant differences (paired t-tests: all outcomes $P>0.05$). This was so when all sites were compared together, and also when sets of sites were compared, upstream versus downstream, within each individual TWINSPAN group.

DISCUSSION

Overall there was good evidence for the existence of different invertebrate communities in streams crossing the WHW, with four main community types identified by the study. However there was no evidence to suggest that this variation, and the variation in water quality which these differences indicate, was associated with impacts that might be associated with recreational use of the long-distance footpath, whether due to associated wild camping, or other activities. In line with findings elsewhere (e.g. Langan & Soulsby, 2001; Soulsby et al., 2002), it is more likely that the variation in biologically-assessed water quality observed in streams along the length of the West Highland Way is associated with differences in the natural and land-use characteristics of the individual catchments feeding each of the streams sampled.

This outcome seems encouraging given the current relatively high visitor usage of the WHWWay, and the results form a baseline against which further monitoring might be undertaken, particularly should recreational use of the West Highland Way continue to grow.

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