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Differences in the Quality of Seepage Water and Runoff Caused by Plant Community and Grazing at an Alpine Site in Hol, Southern Norway

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Abstract Alpine ecosystems, representing a large proportion of the land area in Europe, are under pressure from changes in climate and land-use. This may also impact the quality of drainage waters. Here, we assess effects of plant communities (snowbed, dwarf shrub heath, and tall herb meadow) on concentrations of dissolved organic carbon and nitrogen (DOC and DON), ammonium (NH_4-N), nitrate (NO_3-N), and phosphorus (tot-P and PO_4-P) in locally derived seepage water in a non-fertilized sub-alpine area of southern Norway. In addition, we investigated effects of two density levels of sheep (no sheep and 80 sheep km^{-2}) on infiltration capacity, pore size distribution and concentrations of nutrients and bacteria in surface runoff. Concentrations of NO_3-N (<0.02–0.03 mg l^{-1}) and NH_4-N (<0.02–0.03 mg l^{-1}) were low in seepage waters with no significant differences associated with plant community. Also, concentrations of DOC and DON were low, in particular in snowbeds, probably due to low productivity and small soil carbon pools. Infiltration rates, which were significantly smaller in snowbeds than in tall herb meadow, were further reduced by grazing. In turn, this caused increased runoff of coliform bacteria, whereas no effect of grazing on NH_4-N, NO_3-N and PO_4-P was observed. Grazing may significantly alter biological water quality but is not likely to affect the productivity of surface waters in non-fertilized alpine areas.

Keywords Alpine system · Vegetation · Grazing · Nutrients · Coliforms · Surface runoff

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1 Introduction

High-elevation upland ecosystems are potentially important sources of carbon and nutrients to downstream freshwater and coastal waters. Such ecosystems retain atmospheric pollutants (Elser et al. 2009; Fenn et al. 1998) and are a source of dissolved organic matter (DOM) and associated nutrients and pollutants (Battarbee et al. 2009; Hessen et al. 2009; Hood et al. 2003). The nutrient retention capacity of mountainous regions is subject to a

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number of drivers of change, such as climate (Beniston 2009), land-use (Körner et al. 2005), and N deposition (Hole and Engardt 2008). Despite their considerable extension, little is known about the role of alpine ecosystems on surface water quality.

The vegetation cover in alpine areas consists of a mosaic of different plant communities determined by climatic, topographical, lithological, and edaphic factors as well as land-use (Körner 2003; Nagy and Grabherr 2009). The composition of the vegetation and its spatial distribution, both strongly affected by edaphic factors, may modify biogeochemical properties and processes in soils (Nielsen et al. 2009; Strand et al. 2008; Vinton and Burke 1997) important for surface water quality (Palmer et al. 2001; Sjøeng et al. 2007; Strand et al. 2008).

DOM was reported to be lower in surface water from non-forested than from forested catchments (Hood et al. 2005; Hood et al. 2003; Skjelkvåle and Wright 1998) and was positively related to carbon (C) and nitrogen (N) contents, and C/N ratios of the soil (Hood et al. 2003; Strand et al. 2008). By contrast, concentrations of NO_3 and NH_4 were negatively related to C content and C/N ratio of the soil (Hood et al. 2003; Strand et al. 2008). The concentration of NO_3 in Norwegian lakes was positively related to N deposition and negatively to vegetation density in the catchment (Hessen et al. 2009). Furthermore, Sjøeng et al. (2007) found a significant positive correlation between percentage bare rock and amount of NO_3 leached in 12 headwater catchments with a high N deposition (1.63–2.75 g m^{-2} $year^{-1}$) in southwest Norway. These studies clearly highlight the important role of vegetation and soil in controlling leaching of DOM and nutrients from non-fertilized ecosystems to surface waters. Thus, land-use affecting both vegetation and soil may be an important driver for a change in surface water quality.

Grazing by large herbivores may alter vegetation structure (Austrheim and Eriksson 2001; Speed et al. 2010), with potential effects on physical and chemical soil properties and processes (Frank and Groffman 1998; van der Wal et al. 2001). Additionally, grazing also may increase loads of sediments, nutrients, and fecal bacteria in surface runoff due to removal of vegetation and reduced soil infiltration (Derlet et al. 2008; Elliott and Carlson 2004; Meyles et al. 2006; Muirhead et al. 2006). In a small catchment at Dartmoor, UK, Meyles et al. (2006) found clear effects of sheep grazing on bulk density (increased), porosity (reduced) and soil water content at standard matrix pressures (reduced). They argue that grazing, even without removing the vegetation completely, may reduce the soil's wetness threshold (i.e., field capacity is reached more rapidly) at intensively grazed sites, causing a more rapid soil water movement to streams (Meyles et al. 2006).

In Norway, sheep are the most important large herbivore grazing in natural and semi-natural habitats. Although sheep grazing pressure in Norway declined by 5 % from 1949 to 1999 the proportion of sheep grazing in mountain areas increased by 7 % (Austrheim et al. 2011). Sheep grazing is also an important land-use in the north-Atlantic region of Europe (UK, Iceland, and Faroe Islands) as well in several Mediterranean countries (Dýrmundsson 2006; Hadjigeorgiou et al. 2005). So far, most studies on effects of herbivores on water quality in mountain areas have been done in intensively managed, fertilized systems, whereas detailed investigations in non-fertilized semi-natural alpine areas are rare.

The objectives of this study were to assess (1) the quality of locally derived seepage water (DOC, DON, NH_4-N, NO_3-N, and PO_4-P) in three non-fertilized alpine plant communities of increasing productivity (snowbed<dwarf shrub heath<tall herb meadow) at Hol, southern Norway, (2) effects of two density levels of sheep (no sheep and high sheep density; 80 sheep km^2) and plant community (snowbed and tall herb meadow) on infiltration capacity and pore size distribution, and (3) the impact of grazing (no sheep and high sheep density) on streamwater quality (nutrients and fecal contamination).

We hypothesize the concentration of DOC and DON in locally derived seepage water to increase and inorganic N concentration to decrease with increased vegetation productivity, due to larger pools of soil organic matter (SOM) and greater N demands in higher productivity vegetation types (*H1*). We also hypothesize infiltration rate and fraction of macropores to be smaller in snowbeds vs. tall herb meadow and smaller in the grazed compared to not grazed area (*H2*). As a direct measure of sheep impact on surface water quality, we hypothesize the concentration of inorganic N and bacteria [i.e., total coliforms and *Escherichia coli* (*E. coli*)] to be greater in grazed vs. not grazed catchment (*H3*).

2 Material and Methods

2.1 Site Description

The study site is located in the low-alpine region (1,050–1,320 m a.s.l.) in Hol municipality, Buskerud county, southern Norway (7°55′–8°00′ E, 60°40′–60°45′ N). The site is within a large-fenced enclosure (~2.7 km^{-2}, Fig. 1), established in 2001 as part of a controlled grazing experiment (Mysterud and Austrheim 2005). The fenced enclosure is divided into three blocks, each with tree sub-enclosures (approximately 0.3 km^2 each) with no sheep (control), low density (25 sheep km^{-2}) and high density (80 sheep km^{-2}) of domestic sheep (*Ovis aries*). Sheep grazing occurs from the end of June to the beginning of September each year (Mysterud and Austrheim 2005). The bedrock consists of meta-arkose and quaternary deposits of till and colluvium (Kristiansen and Sollid 1985; Sigmond 1998). Soils are acidic and spatially variable, including peaty deposits in topographic depressions and freely drained soils with shallow organic horizons (Martinsen et al. 2011). Vegetation is dominated by dwarf shrub heaths with smaller patches of lichen heaths, snow beds and alpine meadow communities at lee-sides (Rekdal 2001b). Mean annual temperature is −1.5 °C and mean annual precipitation is about 1000 mm (Evju et al. 2009). Temperature and precipitation varies considerable during the growing season (from mid June to mid September) and between years (Online resource 1). The average wet N deposition rate is small and estimated at ~0.42 g m^2 $year^{-1}$ (Aas et al. 2008).

2.2 Location and Sampling Procedures

The most frequent form of snowbed at the study site is sedge and grass snowbed (SNOWB), dominated by *Avenella flexuosa* with elements of *Salix herbacea*, *Carex bigelowii*, *Anthoxanthum oduratum* and *Alchemilla alpina* (Rekdal 2001a; b). Snowbeds are located in areas with depressions leading to a deep and long lasting snow cover,

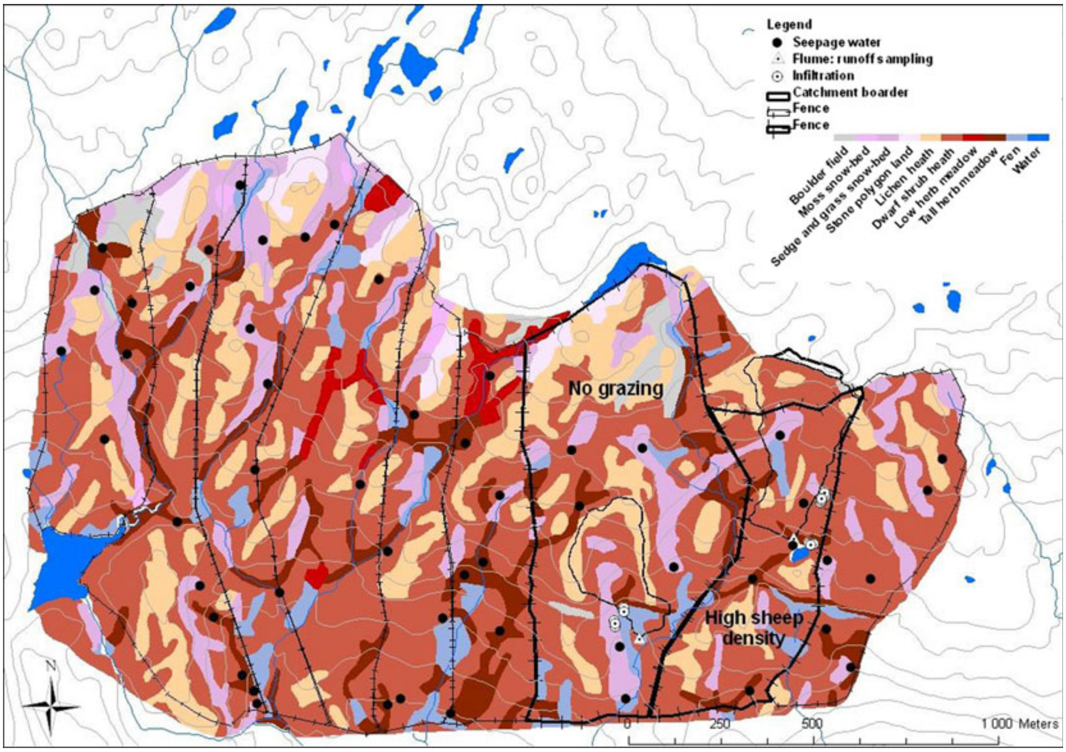

Fig. 1 Sampling locations and experimental area, Hol, Southern Norway. Different plant communities and sites used for water collection and infiltration (pore size determination) measurements are listed in the legend. The map is modified after Rekdal (2001b) using ESRI® ArcMap™ 9.3

on soils of low to intermediate fertility with great variations in water availability in the course of the growing season (Rekdal 2001a). Dwarf shrub heath (DSH), dominated by low shrubs (*Betula nana*, *Vaccinium myrtillus*, *Empetrium nigrum* spp.) and *A. flexuosa* (Rekdal 2001b), occurs in areas with a stable snow cover on soils of low to intermediate fertility with moderate water availability (Rekdal 2001a). Tall herb meadow (THM) is a species rich vegetation community including *Salix* spp., herbs and ferns typical for topographic depressions along streams and rivers, with a stable snow cover, but relatively early snowmelt as compared to snowbeds. THM is located on sites of high fertility with high water availability either from soil- or stream water (Rekdal 2001a). Annual aboveground biomass production (g dry weight m^{-2} $year^{-1}$) increases in order SNOWB (25–75 g m^{-2} $year^{-1}$)<DSH (50–100 g m^{-2} $year^{-1}$)<THM (100–250 g m^{-2} $year^{-1}$) (Rekdal 2001a).

In September 2006, 54 small seeps, with locally derived water after rainfall from within the plant communities SNOWB (n=18), DSH (n=21) and THM (n=15) were selected for sampling (Fig. 1). Seepage water was defined as water percolating the unsaturated zone of the soil mixed with some surface runoff. Due to the limitation in the number of seeps, the three plant communities were not represented in all sub-enclosures. Thus, grazing effects were not included in the statistical analysis. A total of 107 samples of seeps were collected few days after rain events using PVC bottles in August and September 2006 (n=52) and in June to September 2007 (n=55). Samples were stored cold (4 °C) and dark prior to analysis. Analysis of tot-P and PO_4-P were conducted on samples from 2007 only.

Measurement of infiltration rate and pore size distribution were conducted at six different sites, each consisting of three plots (Fig. 1). Two sites were located in an enclosure with no grazing in THM (n=3) and SNOWB (n=3). Four sites were located in an enclosure with high sheep density in THM and SNOWB outside sheep tracks ($n_{\mathrm{THM\ no\ track}}$=3, $n_{\mathrm{SNOWB\ no\ track}}$=3) and on well defined sheep tracks ($n_{\mathrm{THM\ track}}$=3, $n_{\mathrm{SNOWB\ track}}$=3), respectively.

Infiltration rates (in centimeter per hour) were determined as the amount of water per surface area and time penetrating the soils using double (outer ring: Φ 55–58, inner ring: Φ 30–33 cm) and single ring (Φ 30–33 cm) infiltrometers. In most cases, single ring infiltrometers (inserted 7–8 cm into the soil) were used due to installation problems caused by the presence of stones at the soil surface. The soils were pre-wetted for 1 h before the rate measurement in order to saturate the soil pores (i.e., obtain steady infiltration rates), thereby reducing the risk of horizontal flow. Readings of the decreasing water level within the inner rings were taken every 10 min between 1 and 2 h after start of the experiment (i.e., seven records for each plot). For the statistical analysis, the mean of the seven records for each plot was used (n=18) to avoid pseudoreplication. The measurements were conducted with a falling hydraulic head. In case of high infiltration rate, water was added (i.e., increased hydraulic head) to the inner rings.

At each site, two samples of the OA-horizon were taken using 100 cm^3 steel rings to a maximum depth of 3.7 cm (n=36). The undisturbed soil samples were used to determine the distribution of the pore size fractions>200 μm (i.e., macropores), 30–200 μm (i.e., mesopores), and 3–30 μm (i.e., micropores) using ceramic pressure plates (Richards 1948). The different pore size fractions were calculated based on weight of the soil samples at different matrix potentials (pF). The macropores were determined as the difference in weight between water saturation and pF 1 (i.e., −0.02 bar), the mesopores were determined by weight difference between pF 1 and pF 2 (i.e., −0.1 bar) and the micropores as the difference in weight between pF 2 and pF 3 (i.e., - 1 bar). Bulk density (BD; g cm^{-3}) was determined at pF 3 and not at the wilting point (pF 4.2) due to loss of all data at pF 4.2. Thus the reported values represent a slight over-estimation of the true BD. A further description of the procedure is given by Ness (2008). In the statistical analysis, the mean of the two OA-horizons at each site was used for each of the three fractions (n=18).

Only two of the enclosures had a headwater catchment largely within its boundaries. One was situated in the enclosure with no grazing (~5.3 ha) and the other in the enclosure with a high sheep density (~8.9 ha) (Fig. 1, Online resource 2). The percentage of low to intermediate productive plant communities was somewhat smaller (61 %) in the non-grazed as compared to the grazed (72 %) catchment with the percentage fen being similar (Online resource 2). In June 2007, flumes [RBC flume (0.16–9 l s^{-1}), type 13.17.02, (Eijkelkamp 2001)] for water flux measurements were installed in two small streams draining the two catchments (Online resource 3). Streamwater samples at each flume were collected during the growing season. This was from June to September 2007 (n=46), from June to October 2008 (n=24) and from May to September 2009 (n=66).

For sampling, we used PVC bottles for chemical analysis and sterile Polysterol bottles (IDEXX Laboratories) for fecal bacteria determination. The water samples used for chemical analysis were stored dark and cold (4 °C) prior to analysis. Samples used for bacteria determination were analyzed short time after sampling. Analysis of tot-P, PO_4-P, total coliforms, and *E. coli* were conducted on samples from 2009 only.

2.3 Chemical Analysis

Conductivity (712 Conductometer) and pH (Orion, model 720) were determined on unfiltered water samples. All water samples were filtered (0.45 μm) prior to further chemical analysis. Nitrate-N (sum of NO_3^- and NO_2^-) was determined photometrically (flow injection analysis; FIA star 5020 analyzer, Tecator) according to the Norwegian standard NS 4745 (NSF 1975a). Ammonium-N was determined photometrically (Photometer, Gilford Instrument) according to (NSF 1975b). Total-N was determined photometrically (flow injection analysis; FIA star 5020 analyzer, Tecator) after oxidation by peroxodisulfate according to the Norwegian standard NS 4743 (NSF 1993). Dissolved organic N (DON) was calculated as total-N less the sum of NH_4-N and NO_3-N. Dissolved organic carbon (DOC) was measured using a total organic carbon analyzer (TOC-V CPN, Shimadzu) according to NS 1484 (NSF 1997b). The detection limit was 0.02 mg l^{-1} for NO_3-N, NH_4-N and total-N and 0.2 mg l^{-1} for DOC. Total P (after oxidation with potassium peroxodisulfate to orthophosphate) and PO_4-P were determined photometrically (UV-1201 UV–VIS Spectrophotometer, Shimadzu) after reaction with ammonium molybdate according to the Norwegian standard NS-EN 1189 (NSF 1997a). The detection limit for phosphorous was 1 μg l^{-1}.

The determination and quantification of waterborne bacteria, i.e., total coliforms and *E. coli*, were conducted on unfiltered water samples using Colilert®-18 and Quanty-Tray®/2000 (IDEXX Laboratories Inc.) according to the procedure described by Eckner (1998). The microbiological analyses were conducted in surface runoff only (i.e., not in the seeps). Quanty-Tray®/2000 is a semi-automated quantification method based on the standard most probable number (MPN) of bacteria in a water sample providing counts from 1 $(100\ ml)^{-1}$ to a maximum of 2419.6 $(100\ ml)^{-1}$ without dilution (IDEXX Laboratories Inc. 2010). Water samples collected after 31/07/09 were diluted (streamwater/distilled water 1:1). For statistical purposes, we used two times the upper detection limit for samples with counts>2,419.6 MPN 100 ml^{-1} (i.e., 4839.2 for undiluted samples and 2*4839.2=9678.4 for diluted samples). Half the detection limit was used for samples with counts<1 MPN 100 ml^{-1} (i.e., 0.5 for undiluted and 2×0.5=1 for diluted samples). A further description of the procedure is given by Grund (2010).

2.4 Statistical Analysis

One-way analysis of variance was used to determine (1) differences in the quality parameters of seep water between plant communities (DSH, SNOWB and THM; three levels) and (2) differences in BD, infiltration rate and pore size distribution between grazing treatment, plant community and sheep track (CSNOWB=control (no grazing) snowbed, CTHM=control tall herb meadow, HSNOWB=high sheep density snowbed, HTHM=high sheep density tall herb meadow, HSNOWBtr=high sheep density snowbed on sheep track and HTHMtr=high sheep density tall herb meadow on sheep track; six levels). Analysis of co-variance was used to test for the relationship between the concentration of NH_4-N, total coliforms and *E. coli* on the one hand and water flux (Q; continuous explanatory variable) and sheep density (no grazing and high sheep density; categorical explanatory variable), on the other. Values below the detection limit were not included in the statistical analysis. Some variables were ln or sqrt transformed prior to analysis to avoid violations of the model assumptions. Residuals were plotted (histograms and QQ normal plots) to assess normality and potential outliers. Model simplification was conducted by stepwise deletion of non-significant terms (Online resource 4).

Differences in quality parameters of runoff in the two catchments (Fig. 1) with no grazing and a high sheep density were assessed using two sided *t* tests. The statistical software package "R", version 2.13.2 (R Development Core Team 2011), was used for all statistical analyses. All analyses were also conducted using linear mixed effects models to account for spatial and temporal variation in the data. The random effects included were year of sampling and block for seepage water parameters, month of sampling for runoff quality parameters and record (seven levels) for infiltration measurements. However, the results of the linear mixed effects models did not differ substantially from the ones described above and resulted in unnecessarily complex models. Thus, these are not included in the presentation.

3 Results

3.1 Chemistry of Locally Derived Seepage Water

Conductivity (in microsiemens per centimeter) and concentrations (in milligram per liter) of total-N, DON, and DOC in seepage water (Fig. 1) depended significantly on the plant community (Table 1). Conductivity was significantly greater in THM as compared to DSH and SNOWB communities. DOC, total-N, and DON concentrations were significantly smaller in SNOWB (~1.09, 0.05, and 0.05 mg l^{-1}, respectively) than in DSH (~1.96, 0.08, and 0.07 mg l^{-1}, respectively) and THM (~1.50, 0.06, and 0.06 mg l^{-1}, respectively). The concentration of NO_3-N was below the detection limit for all the samples and NH_4-N was only present in ten of the 107 samples. Of these, eight were sampled in DSH (Table 1). Total-N largely consisted of DON due to the low concentrations of inorganic N. The pH and DOC/DON ratio did not differ significantly between the plant communities. The concentration of total P and PO_4-P were at or below the detection limit (1 μg l^{-1}) in seepage water from all systems.

3.2 Effects of Plant Community and Grazing on Infiltration and Pore Size Distribution

Bulk density (BD; in gram per cubic centimeter) was significantly greater in SNOWB than in THM (Fig. 2a), whereas the infiltration rate (in centimeter per hour) was significantly smaller in SNOWB (Fig. 2b). In THM, but not in SNOWB, grazing increased BD significantly (Fig. 2a). However, the effect of grazing on infiltration rate was not significant in any of the two plant communities. Not surprisingly, sheep tracks had significantly smaller infiltration rates than the areas outside the tracks in both SNOWB and THM, due to soil compaction caused by heavy sheep traffic. The relative volume of macropores (>200 μm) was significantly greater in THM than in SNOWB at the non-grazed site (~20 % vs. ~10 %) but differed only slightly at the grazed site (~10 % vs. ~8.5 %) (Fig. 2c). The relative volume of macropores in soils of the sheep tracks was significantly smaller than outside the tracks in both plant communities. The smaller volume

Table 1 Chemical characteristics of seepage water within three plant communities in an alpine system, Hol, Southern Norway

Variable	SNOWB			DSH			THM		
	Mean	se	*n*	Mean	se	*n*	Mean	se	*n*
Conductivity (μS cm^{-1})	11.9 A	1.3	35	13.7 A	1.1	36	16.9 B	1.1	34
PH	6.5 A	0.1	33	6.7 A	0.1	36	6.7 A	0.1	35
NO_3^-N (mg l^{-1})[a]	–	–	–	–	–	–	–	–	–
NH_4–N (mg l^{-1})[a]	0.03–	–	1	0.03–	0.01	8	0.03–	–	1
Tot-N (mg l^{-1})	0.05 A	0.005	30	0.08 B	0.008	35	0.06 B	0.006	34
DON (mg l^{-1})	0.05 A	0.005	30	0.07 B	0.008	35	0.06 B	0.006	34
DOC (mg l^{-1})	1.09 A	0.118	35	1.96 B	0.247	36	1.50 B	0.129	36
DOC/DON ratio	29.8 A	3.1	30	28.7 A	1.9	35	27.7 A	1.3	34

SNOWB snow bed, *DSH* dwarf shrub heath, *THM* tall herb meadow. Mean values, standard error (se) and number of samples (*n*) is shown. "–" indicates no value. Different capital letters next to the mean values indicate difference between plant communities for the selected variables at a level of significance $p<0.05$ (one-way ANOVA)

Conductivity, DOC and DOC/DON were ln transformed and tot-N and DON sqrt transformed prior to analysis. One value of the measured conductivity (THM, 83 μS cm^{-1}) was considered as outlier and thus omitted from the analysis. This did not affect the outcome of the tests

[a] All samples out of 93 analyzed for NO_3-N and 97 out of 107 samples analyzed for NH_4-N were below the detection limit

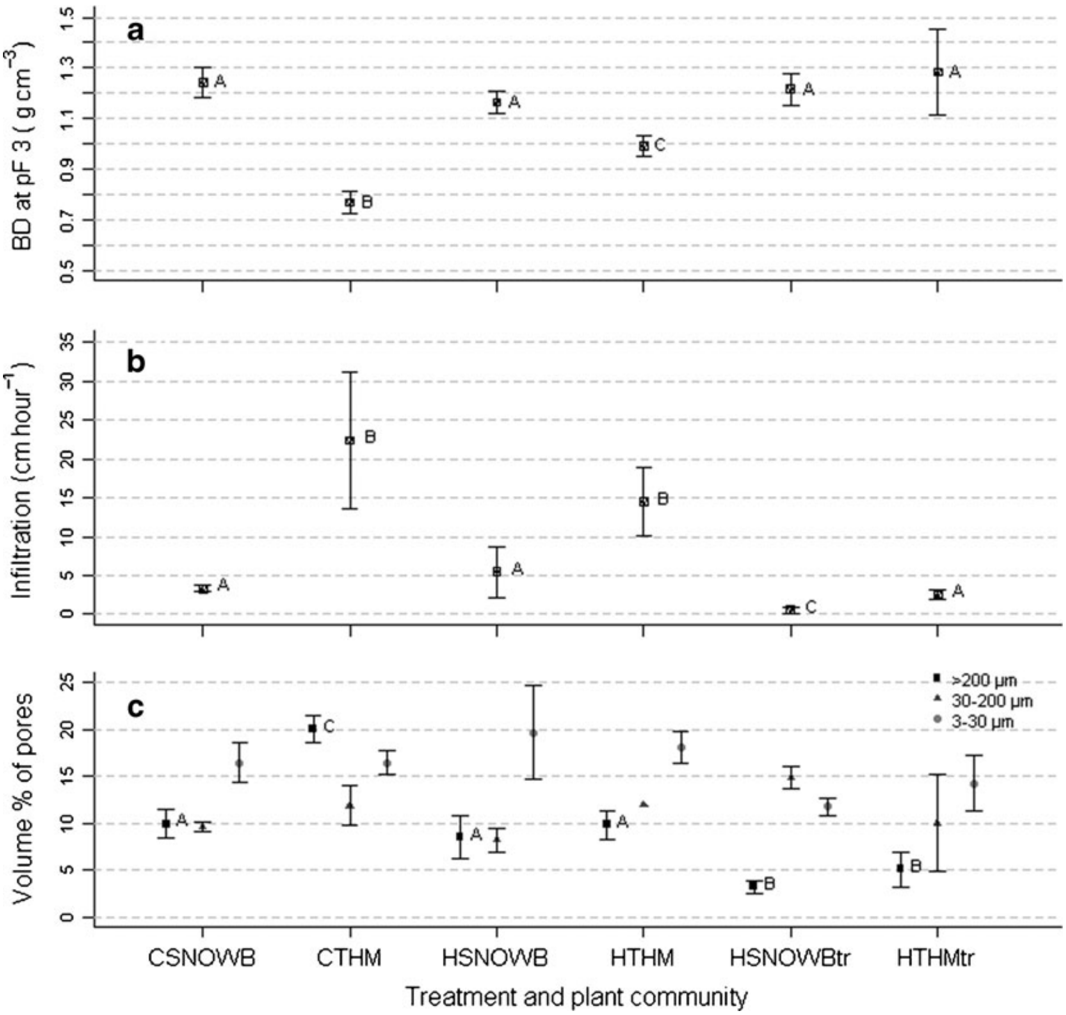

Fig. 2 Effects of sheep grazing (no grazing and high sheep density) and plant community (snowbed and tall herb meadow) and trampling (sheep tracks vs no tracks) on **a** mean (±se) bulk density (BD), **b** infiltration rates (in centimeter per hour), and **c** pore size distribution (% of the total pore volume) of the pore size fractions >200 μm ("macropores"), 30–200 μm ("mesopores"), and 3–30 μm ("micropores") of OA-horizon soils, Hol, Southern Norway. Different capital letters indicate difference between treatment combinations at a level of significance $p<0.05$ (one-way ANOVA; infiltration rates and BD were ln-transformed prior to analysis). No letters are indicated for the mesopores and micropore fractions which did not differ significantly between the treatments. CSNOWB=no grazing in snowbed, CTHM=no grazing in tall herb meadow, *HSNOWB* high sheep density in snowbed, *HTHM* high sheep density in tall herb meadow. HSNOWtr and HTMtr is high sheep density on tracks in snowbed and tall herb meadow, respectively. $n=3$ for each treatment combination except for the mesopores fraction at HTHM with 1 observation and HSNOWB with two observations

percentage of macropores in soils on tracks as compared to no tracks was in accordance with their smaller infiltration rate and greater BD (in THM only). The relative volume of mesopores (30–200 μm) and micropores (3–30 μm) were not significantly affected by plant community, grazing or tracks (Fig. 2c).

3.3 Effects of Grazing on Chemical and Biological Quality Parameters of Runoff

Concentrations of tot-N, DON, and DOC were in general larger and associated with a greater variation in the non-grazed catchment (Table 2). Also conductivity and

Table 2 Mean chemical and biological quality parameters of surface runoff from two catchments with no grazing and a high sheep density (80 sheep km^{-2}) in an alpine system, Hol, Southern Norway

Variable	No grazing			High sheep density		
	Mean	se	*n*	Mean	se	*n*
Conductivity ($\mu S\ cm^{-1}$)	33.0 A	1.9	54	8.3 B	0.2	70
pH	7.3 A	0.1	54	6.7 B	0.0	70
NO_3-N (mg l^{-1})[a]	0.02 A	0.00	3	0.03 A	0.01	3
NH_4-N (mg l^{-1})[a]	0.02 A	0.00	8	0 03 A	0.00	8
Tot-N (mg l^{-1})	0.12 A	0.01	57	0.08 B	0.00	73
DON (mg l^{-1})	0.12 A	0.01	57	0.07 B	0.00	73
DOC (mg l^{-1})	2.80 A	0.13	59	1.88 B	0.05	75
DOC/DON ratio	26.7 A	1.1	56	28.3 A	1.1	72
PO_4-P ($\mu g\ l^{-1}$)	1.39 A	0.07	8	1.48 A	0.13	14
Tot-P ($\mu g\ l^{-1}$)	1.71 A	0.26	29	2.01 A	0.20	34
Total coliform (MPN 100 ml^{-1})	704 A	239	16	2,109 B	564	19
E. coli (MPN 100 ml^{-1})	10 A	3	17	81 B	33	20

Standard error (se) and number of samples (*n*) is shown. Different capital letters indicate difference between the catchments for selected variables at a level of significance $p<0.05$ (two-sided *t* tests)

[a] Eighty two samples out of 88 analyzed for NO_3-N and 114 out of 130 samples analyzed for NH_4-N were below the detection limit

pH were larger in the catchment without sheep. As observed in seepage water (Table 1), also in catchment runoff DON was the major constituent of total-N. The concentration of NH_4-N and NO_3-N in runoff did not differ significantly between the catchments (without sheep: 0.02 and 0.02 mg l^{-1}, respectively and at high sheep density: 0.03 and 0.03 mg l^{-1}, respectively). This indicated little effect of grazing on the leaching of inorganic N. By contrast, the concentration of total coliform bacteria and *E. coli* (expressed as the MPN 100 ml^{-1}) were significantly greater in the grazed (2109 and 81, respectively) than in the non-grazed (704 and 10, respectively) catchment. Concentrations of PO_4-P and total P were small, but greater than detection limit, and did not differ significantly between the catchments (Table 2).

The concentration of NH_4-N was not related to Q in any of the catchments (p=0.189). Neither was there a significant effect of Q on total coliforms (p=0.502). However, the amount of *E. coli*, being a strong indicator of fecal contamination, was significantly explained by Q and catchment [$p<0.001$, R^2=0.62 (sqrt-transformed and two outliers removed] with a significantly greater positive response ($p<0.001$) at the grazed as compared to the non-grazed catchment (Online resource 4).

4 Discussion

Clean water is a crucial ecosystem service provided by alpine ecosystems globally. Nearly half the human population are depending on water supply from mountains (Körner et al. 2005). Yet there is limited information on how factors like plant communities and livestock grazing affect physical soil properties and water quality. This study at Hol shows that bulk density was significantly greater in the SNOWB as compared to the THM (Fig. 2a). This is in accordance with previous findings at the study site (Martinsen et al. 2011) and probably reflects the greater snowpack (hence compaction) in SNOWB. We found no difference in BD between the two grazing treatments in the SNOWB, which suggests a greater effect of snow cover than of trampling. By contrast, within the THM, BD was significantly greater at the grazed as compared to the non-grazed site (Fig. 2a). As THM is classified with a higher grazing value compared to snowbed (Mobæk et al. 2009) and the sheep spend much time here while feeding, a larger impact of grazing is to be expected. The grazing-induced increase in BD associated with a significant decrease in infiltration rate, is in accordance with other studies [e.g., Steffens et al. (2008) and Wheeler et al. (2002)].

Infiltration rates differed considerably between the soils of the plant communities and between the no sheep tracks vs. the well defined tracks (Fig. 2b). The smallest infiltration rate (0.54 cm h^{-1}) was observed in SNOWB on a well defined sheep track (Fig. 2b). The amount of precipitation (sum per hour) exceeded this amount on several occasions during the years from 2007 to 2009 (Online resource 3). As infiltration rates determined by ring infiltrometers tend to overestimate the true vertical infiltration capacity (~20 % for rings of diameter 30 cm) (Tricker 1978), substantial amounts of surface runoff must have been generated in the sheep tracks.

Plant community and grazing had greater effects on the macropore size fraction (i.e., >200 μm) than on the fractions of meso- (30–200 μm) and micropores (3–30 μm) (Fig. 2c). The smaller fraction of macropores in the sheep tracks, were in accordance with the smaller infiltration rate and greater BD (in THM only) as compared to outside the tracks. The small effects of plant community and grazing (outside the tracks) on the meso- and micropore size fractions in soils were not surprising, as disruption of these fractions requires a substantial pressure (Jansson and Johansson 1998). They found only minor changes in the pore size fractions 10–50 μm after compaction by a tracked and wheeled forest machine (about 20,000 kg).

Seepage water had low concentrations of both NO_3-N and NH_4-N that did not differ between the plant communities, despite differences in soil fertility (Rekdal 2001a). The concentration of NO_3-N never even exceeded the detection limit (Table 1). This is in line with in situ concentrations of NO_3-N and NH_4-N in O-horizon water, which were low in both grazed and non-grazed enclosures (0.06 mg l^{-1} and 0.06 mg l^{-1}, respectively; results not shown). Earlier, low concentrations of NH_4-N (~0.005-0.03 mg l^{-1}) in O-horizon water of soils at Zackenberg in Northeast Greenland were reported by Elberling et al. (2008). The concentrations of NH_4-N and NO_3-N in streamwater (Table 2) were in the same low range as those in seepage water (Table 1). This indicates a strong demand for inorganic N and a small net nitrification potential in the low-alpine ecosystem at Hol, irrespective of grazing density. This is further supported by the previously reported small rates of in situ net-mineralization and nitrification in O-horizons of grassland habitats at Hol (Martinsen et al. 2012). Similar to inorganic N, also the concentration of PO_4-P was small in seepage water from all systems, as well as in catchment runoff.

Runoff of inorganic N tends to increase in areas with elevated atmospheric N deposition (Hessen et al. 2009) and is often interpreted as a first sign of N-saturation. At Hol, with a relatively small estimated N deposition rate (0.42 g m^2 $year^{-1}$; Aas et al. 2008), total-N in seepage and stream water was dominated by DON (Tables 1 and 2). Particularly in N-saturated catchments, it is commonly found that N retention is seasonally variable with higher N in runoff during winter and snowmelt (de Wit et al. 2008). However, at Hol, two stream water samples collected in October 2008 and four collected during snowmelt in May 2009, did not show elevated N runoff in autumn and early spring (concentrations of NH_4-N and NO_3-N were below the detection limit, viz. 0.02 mg l^{-1}). Thus, grazing-induced effects on potential N mineralization as reported by Martinsen et al. (2012), which could increase leaching of NO_3-N (McNeill and Unkovich 2007), were not reflected by the seepage—and stream water chemistry in this system. Effective retention of N in soil and vegetation (Sjøeng et al. 2007) may explain the low inorganic N concentrations observed at our study site.

Concentrations of DOC in seepage water (means 1.09–1.96 mg l^{-1}) and in runoff from the two catchments (means 1.88 and 2.80 mg l^{-1}) were at the lower end of the range of values reported by Sjöeng et al. (2007) at Måkevatn and Øygard, southern Norway (2.4–4.2 mg l^{-1} and 1.2–7.5 mg l^{-1} for the Måkevatn and Øygard catchments, respectively) and by Skjelkvåle and Wright (1998). In a Norwegian lake survey, Skjelkvåle and Wright (1998) found TOC concentrations of 0.4 to 4.3 mg l^{-1} for mountain lakes and lakes in forested catchments, respectively. However, our values were smaller than TOC concentrations reported by Strand et al. (2008) in runoff at Storgama (5.6–21.9 mg l^{-1}) and by Wright and Jenkins (2001) at Risdalsheia, southern Norway (7.2–18 mg l^{-1}).

In terms of fluxes, assuming that yearly runoff was only slightly less than precipitation (i.e., ca 1,000 mm $year^{-1}$) and no variation occurred in DOC concentration between growing season and winter, DOC fluxes were approximately 1.1, 1.5, and 2 g m^{-2} $year^{-1}$ for SNOWB, THM and DSH, respectively. These fluxes were rather small as compared to those reported for e.g. forested catchments in Finland [(3–10 g m^{-2} $year^{-1}$ (Mattsson et al. 2003);

2.3–14.8 g m^{-2} $year^{-1}$ (Rantakari et al. 2010)] but in the same range as fluxes of TOC reported by Wright et al. (1994) from alpine catchments in Sogndal, Norway (0.65 to 2.1 g m^{-2} $year^{-1}$).

Concentrations of DOC and DON in seepage water increased in the order SNOWB<THM<DSH. The differences are in accordance with previous reports on biomass production (Rekdal 2001a) and soil C-pools (Martinsen et al. 2011), both being greater in THM as compared to SNOWB. As DOC is positively related to the content of SOM (Strand et al. 2008), we argue that the observed differences most likely reflect differences associated with the plant communities and primary production rates. Because the fraction of the most productive plant communities (DSH and THM) was somewhat greater in the grazed (67 %) as compared to the non-grazed (54 %) catchment, while the percentage fen was similar (Online resource 2), we expected greater DOC concentrations in the former. This was not confirmed by our observations, which showed a significantly smaller concentration of DOC in the runoff water of the grazed catchment (Table 2). Previously, it was shown that DOC in runoff water also depends on the fraction of wetlands (Hope et al. 1997; Laudon et al. 2004; Rantakari et al. 2010) and their location within a catchment (Billett et al. 2006). Sub-catchment analysis of soil C-pools and export of DOC revealed a strong relationship between the fraction of peat and concentrations of DOC in the upper part of a catchment in NE Scotland. By contrast, this was not observed further downstream, where the catchment was more influenced by freely draining mineral soils (Billett et al. 2006). At Hol, the location of the fen in the non-grazed catchment was close to the flume where the samples were taken (Fig. 1) suggesting that the location of the fen had a greater impact on export of DOC than did the vegetation distribution of the non-grazed catchment.

Concentrations of total coliforms and *E. coli* were significantly greater in the grazed as compared to the non-grazed catchment (Table 2). We also found a significantly greater increase in concentration of *E. coli* with increasing water flux at the grazed as compared to the non-grazed site, indicating a greater potential for surface water contamination at the latter. Increased concentrations of bacteria with increased discharge are also reported by Collins et al. (2005) and Muirhead et al. (2006). The greater concentrations of total coliforms and *E. coli* in stream water at high discharge suggest an increase in the contribution of surface runoff relatively high in these bacteria. *E. coli* is effectively retained (filtration and adsorption) when infiltrating the soil (Muirhead et al. 2006). The increased concentration of pathogens in runoff with discharge in the grazed catchment is in accordance with its smaller infiltration rate especially in the tracks. In a previous study, a significant amount of coliforms and *E. coli* were found in two seeps in THM in the high sheep density enclosure (Fig. 1). This supports that also seeps may become contaminated by surface runoff at high sheep density (Ness 2008). Despite a strong indication of increased surface runoff in response to grazing, we observed no difference in concentrations of inorganic N and P in stream water. This suggests a relatively strong retention of these nutrients, even in sheep tracks.

5 Conclusions

Plant communities affect the quality of seepage water with greater concentrations of DOC and DON in tall her meadow (productive) as compared to snowbed (less productive). Furthermore, plant community and grazing level are associated with significantly different soil physical properties. Infiltration rates and the fractions of macropores were smaller in snowbeds as compared to tall herb meadows, and a further decline occurs due to heavy sheep traffic. Amounts of coliform bacteria were significantly greater in runoff from the grazed vs. the non-grazed catchment. Probably, this is caused by effect of trampling on soil compaction resulting in increased surface runoff to the stream. Despite the grazing-induced contamination of stream water with coliform bacteria, there is no effect of grazing on inorganic N and P concentrations in stream water. Overall, our results suggest a strong retention of N and P in this low-alpine ecosystem.

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