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Dynamics of abiotic parameters, solute removal and sediment retention in summer-dry headwater streams of western Oregon

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Abstract

Summer-dry headwater streams provide an important interface between aquatic and terrestrial environments. Six summer-dry streams differing in flow duration and exposure were studied in western Oregon. On a temporal and a spatial scale, nitrate patterns in such systems reflect the close connection to subsurface flow and nitrification/denitrification processes in the soil. Retention efficiency for sediment generated from a forest road was high. In ephemeral streams, 60–80% of suspended sediment (1.6 μm $<$ suspended sediment $<$ 53 μm) was removed from the water column over a 75 m stretch at moderate input levels. During injection trials solute removal was largely due to groundwater exchange. Exchange rates between stream water and subsurface flow were estimated at 0.75 and 0.8% per meter of channel. Particularly high removal of nitrate in a meadow stream indicated biological uptake.

Introduction

Headwater streams represent the maximum interface with the terrestrial environment (Vannote et al., 1980). In relative terms, the exchange surface and the biologically active boundary layer between sediment and water column are large. At the same time the density of retention devices in small streams is high (Bilby & Likens, 1980) while stream power is low (Naiman & Sedell, 1979). Large exchange surface and long particle residence time result in short nutrient turnover lengths in headwaters (Minshall et al., 1983; Naiman et al., 1987; Stream Solute Workshop, 1990).

Streams with a recurrent dry phase of varying length are a common feature of the landscape in many parts of the world (Williams, 1996). In temperate regions drought is mostly seasonal and occurs during the summer months. Summer droughts may be a result of decreased precipitation or increased evaporation and evapotranspiration when deciduous leaves are regenerated in spring. During their period of flow, summer-dry headwaters become an integral part of a catchment's drainage system often exceeding permanently wet channels in total length. The biological and physical processes in these small channels then affect the habitat and water quality in permanent downstream reaches, in lentic systems into which many streams discharge, and in groundwaters to which they are connected.

As soils become saturated, stream length in headwater basins will expand (Hewlett & Hibbert, 1967). Rewetting of formerly dry channels results from interflow being intercepted and from a general rise in the groundwater table and subsequent re-activation of discharge areas (*sensu* Hynes, 1983). In many respects water quality in summer-dry headwaters therefore mirrors groundwater qualities and thus geological conditions and soil properties in small drainage areas.

In 1987 we initiated an extensive research project on summer-dry streams in western Oregon. The project was designed to assess the habitat and modifier functions (i.e. the potential to modify habitat quality elsewhere) of these streams. The terminology applied to streams that cease to flow for parts of the year is not consistent (Comin & Williams, 1994). We follow the terminology suggested by Legier & Talin

(1973) to distinguish between ephemeral and temporary streams. Temporary streams have continuous flow for at least 4 months. Flow in ephemeral channels is restricted to post-precipitation periods and may only last for several days. Temporary and ephemeral streams lack a permanent lotic headwater section which characterizes intermittent streams (only parts dry) (Delucchi & Peckarsky, 1989).

In this paper we report on the seasonal dynamics of physical and chemical variables in a permanent, 3 temporary and 3 ephemeral streams. We used injection experiments to study nitrate retention in temporary streams differing in exposure (forest vs. meadow), and road-generated sediment to study particle removal from the water column in ephemeral channels. Knowledge of retention mechanisms and removal capacities provides a baseline for assessing the potential of temporary and ephemeral streams to contribute to water- and habitat-quality in permanent downstream reaches.

Study area and sites

The study sites were located in, and adjacent to McDonald Forest, near Corvallis, Oregon (Figure 1). Most of the field sites were within 5 km of the laboratory. Research efforts therefore could be geared towards key events such as flow initiation, spates, or unseasonally early droughts.

McDonald Forest, property of Oregon State University, is part of the eastern foothills of the Oregon Coast Range. The climate of the Pacific Northwest is characterized by a distinct seasonality. Summers are warm and dry, winters are mild and wet. For Corvallis, Franklin & Dyrness (1973) gave the average July temperature as 18.9 °C (x max 27.1 °C), and January average as 4.0 °C (x min 0.6 °C). Mean annual rainfall is 1000 mm, with less than 50 mm falling from June to August. Because of the distinct seasonality of the rainfall, summer-dry streams are a common feature of the western Oregon landscape.

The characteristic vegetation of the study area is coniferous forest, primarily Douglas fir [*Pseudotsuga menziesii* (Mirb.) Franco]. However, deciduous trees, especially bigleaf maple (*Acer macrophyllum* Pursh), and shrubs are common along small headwaters.

Regular samples were obtained from seven streams. Five of these, including a permanent headwater, were located in the forest, and two were in open meadow areas. Besides exposure, summer-dry streams differed with respect to duration of flow: three were temporary and three were ephemeral. Drainage area for temporary streams was <1 km^2 and <0.25 km^2 for the ephemeral sites. A stream was added to the original study design to assess sediment retention after a forest road was built and used by logging trucks. The stream nomenclature reflects the differences in exposure and flow duration described above (Table 1).

Table 1. Nomenclature of the study sites near Corvallis, Oregon. Names reflect location (exposure) and flow regime

Location	Flow regime	Abbreviation
Forest	**P**ermanent	**PERM**
Forest	**T**emporary	**FT1** and **FT2**
Forest	**E**phemeral	**FE1**, **FE2** and **FE3**
Meadow	**T**emporary	**MT1**
Meadow	**E**phemeral	**ME1**

Study reaches of 160 m in temporary streams and 80 m in ephemeral channels were selected. Within each reach, ten sample sites (2 m of channel) were chosen mainly to study the biota. Five of these sites were selected randomly. Another five were added so that the following criteria were met for each stream: (1) sites with maximum and minimum duration of flow were included; and (2) riffles and pools were equally represented.

Materials and methods

The channels were marked off in 5 m segments prior to the resumption of flow. Stream width was measured for each interval. Slope was obtained separately for each segment. Stream discharge was determined at culverts or v-notch weirs installed to facilitate measurements. Determinations were based on the average of 5 timed collections into a bucket. Current velocity was measured at each sample site with a Montedoro–Whitney flow-meter. The channel bed at each sample site was characterized by determining percent cover of clay, rock, wood and moss (visual estimate). Using segment markers as reference points, drying patterns were recorded to the meter at 2–3 day intervals.

Samples were obtained daily during a logging operation to estimate sediment input from the new forest road and its retention in FE1 and FE3. Water was collected from the unimpacted channel above the road and at the culvert immediately below the road. Serial samples were obtained from both channels at

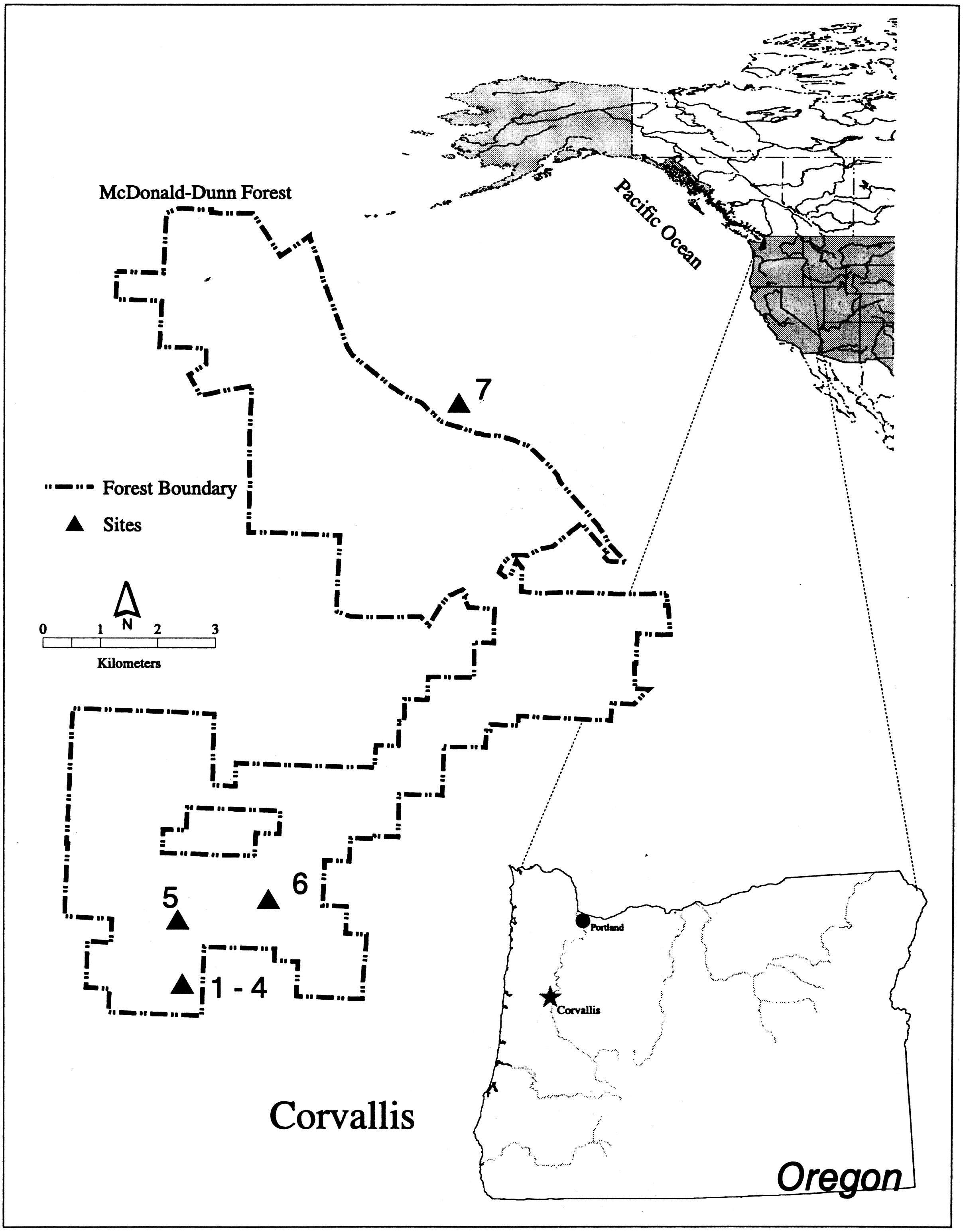

Figure 1. Study area and study sites near Corvallis, Oregon

0 m, 25 m, 50 m, 75 m and 100 m below the culvert to gauge sediment removal. As small trickles draining the road entered the channel just below the uppermost sampling point, values obtained for 0 m in FE1 had to be discarded.

To determine sediment content, the water was filtered through a 53 μm USA Standard Testing Sieve. Material retained in this sieve is defined as non-suspended sediment. Five subsamples of different volumes (10–250 ml) were then taken from the filtered water. These were washed through a pre-weighed Whatman GF/A Glass Microfibre Filter (pore size 1.6 μm). Sediment retained on the filter represents the suspended sediment fraction (53 μm > suspended sediment > 1.6 μm). The non-filterable residue, comprising small inorganic and organic particles as well as solutes is referred to as ultra-fine material. Non-suspended sediment and filters loaded with suspended sediment were oven-dried in aluminum pans for 48 h at 45 °C and then weighed. To quantify amounts of ultra-fine material, three 25 ml aliquots of water that passed through the microfibre filter were evaporated to dryness in pre-weighed pans. The pans were reweighed after the water had evaporated and the non-filtrable residue was determined as the mean increase in pan weight. All weights were obtained on a Mettler electrobalance. Weights are reported as the subsample averages (suspended sediment fraction and ultra-fine material fraction) or as sample totals (non-suspended sediment fraction).

Leaf packs were placed in streams FE1 and FE3 (10 packs FE1, 5 packs FE3) during the period of intense road usage in early spring. Leaf packs in FE1 were part of the regular benthic sampling program, whereas those in FE3 were specifically exposed to monitor sediment deposition. Distances from the road were 63–136 m in FE1 and 12–64 m in FE3. Leaf packs were pulled from the streams after 2 weeks, stored in plastic bags and transported to the laboratory. Packs were then washed over a 53 μm sieve and the wash water containing the suspended sediment was collected. Sediment weight was obtained as described above.

To assess the impact of sediment deposition, core samples taken from a site of high mayfly emergence before and after construction of the forest road were compared. Sample volume was 269.5 cm^3 (38.5 cm^2 $\times$ 7 cm). Both samples were fractionated using a series of Standard Testing Sieves, dried at 45 °C for 72 h and weighed. Organic content for each fraction was determined by loss on ignition (500 °C for 12 h).

Water samples were taken in acid-washed 250 ml polyethylene bottles from all streams, except FE2, to investigate seasonal variation in nitrate concentration. In FE2 flow was modified by re-routing of water from PERM and thus no site-specific samples could be obtained. Samples were stored on ice in the field and at 4 °C in the laboratory. They were filtered through Whatman GF/A Glass Microfibre filters and then analyzed for nitrate within 5 days of sampling. Nitrate concentrations were determined by the Cd-reduction method (APHA, 1985) with a Technicon AutoAnalyzer II.

Solute injection experiments were conducted during April/May 1991 in FT1 and MT1. Nitrate was chosen as the exemplary compound because: (1) it is an important factor in eutrophication caused by land use and atmospheric deposition; (2) losses through volatilization, adsorption on sediment, and transformation driven by the redox potential of the milieu are negligible in oxygen-saturated waters; and (3) cost of analysis at a high sensitivity level is comparatively low.

Study reaches were selected so that discharge at the downstream sampling site (S_D) was equal to, or slightly less than, discharge at the injection site (S_I). A solution containing nitrate (40–45 mg l^{-1}) as the reactive compound as well as rhodamine WT (12 mg l^{-1}) and chloride (1.2 g l^{-1} in FT1 and 6.6 g l^{-1} in MT1) as passive tracers was dripped into the streams for 2 h. Drip rate was kept constant at ca. 100 ml min^{-1} in MT1. Initial drip rate in FT1 was also 100 ml min^{-1}, but it was increased after 40 min to about 150 ml min^{-1}. Water samples were taken at 2.5–20-min intervals about 3 m below the injection site and at a sampling station 25 m downstream in FT1 or 40 m downstream in MT1.

Samples from the injection experiments were treated and analyzed for nitrate as described above. A fluorometer (Turner Model 86) was used to measure rhodamine WT content. Chloride concentration was determined according to the ferric thiocyanate method described by Bergmann & Sanik (1957) and modified by Resche (M. Resche, pers. comm.). This analysis required 125 μl of a saturated mercuric thiocyanate solution and 125 μl of 0.25 m ferric ammonium sulfate to be added to 750 μl of sample solution. The aliquots were well mixed and kept at room temperature for 10 min. Chloride concentrations then were determined at 460 nm in a spectrophotometer.

The uptake length (S_w) for solutes was calculated from the difference in plateau concentrations corrected for background (BG) between S_I and S_D using a log-transformation of the general decay function (Stream Solute Workshop, 1990):

$$S_w = x/(\ln(\text{Solute}_I - \text{BG Solute}_I) - \ln(\text{Solute}_D - \text{BG Solute}_D))$$

where x is the distance between S_I and S_D (25 m in FT1 and 40 m in MT1).

Stream velocity (u) over the injection reach was estimated from the average time for solute concentrations to attain 50% of the downstream plateau level (corrected for background). Only plateau I (P_I) established from the initial drip rate was used to calculate velocity and the velocity-dependent parameter k_c^{-1} (average turnover time of solute molecules) in FT1:

$$k_c^{-1} = S_w u^{-1}$$

Data from the injection experiments also were used to assess catchment properties such as groundwater exchange rates and solute (chloride and nitrate) concentrations in the groundwater pool. Groundwater exchange over the injection reach was assessed by the rate of passive tracer loss (R_T = 1 - $\text{Tracer}_D/\text{Tracer}_I$). Solute levels in the subsurface pools (Solute_{sub}) were then calculated from pre-trial concentrations at the injection and sampling sites:

$$\text{Solute}_{sub} = (\text{Solute}_D - (1 - R_T) * \text{Solute}_I) / R_T$$

Table 2. Physical and chemical parameters characterizing the study streams near Corvallis, Oregon

Parameter	PERM	FT1	FT2	FE1	FE2	MT1	ME1
Elevation [m][1]	165	253	378	197	163	110	186
Width [cm][2]	104	104	94	68	63	46	42
Slope [%][3]	9	11	20	13	12	5	11
Stream bed[4]							
Clay	28	63	15	79	68	11	19
Rock	68	26	69	0	11	87	0
Wood	4	6	4	10	9	0	0
Plants	0	5	12	11	12	2	81
Discharge [$l\ s^{-1}$][5]	57.6	24.7	11.6	5.2	7.2	10.3	1.7
Velocity [$m\ s^{-1}$][6]							
average	–	0.19	0.23	–	–	0.24	–
maximum	–	0.86	0.72	–	–	0.32	–
minimum	–	0.00	0.08	–	–	0.14	–
pH[7]	7.7	7.6	7.3	7.4	7.5	7.3	7.2
Temperature[8]							
average	7.4	8.1	7.9	8.2	7.8	10.1	8.3
maximum	10.0	10.5	10.0	10.5	10.0	17.5	11.0
minimum	4.0	6.0	7.0	6.0	6.0	5.5	6.5

[1] Elevation is given for the mid-study reach.
[2] Measured for the active channel bed in 5 m intervals along each study reach.
[3] Average over the total study reach.
[4] % cover.
[5] Maximum discharge values.
[6] Measured at each sampling site during a single afternoon (13/03/91).
[7] Average measurements cover period from 06/12/89 to 26/07/90.
[8] Based on 15 measurements for long-flow streams and 10 measurements for short-flow streams during the 1989–90 flow period (24/11/89 to 28/04/90).

Results

Stream characteristics

After the summer drought the catchments required ca. 200 mm of almost continuous precipitation before flow resumed in late fall. Maximum discharge was higher in temporary streams as compared with the ephemeral sites but did not exceed 25 $l\ s^{-1}$ (Table 2). Mean velocity for all streams was similar despite considerable differences in slope. Channel slope in the forest streams translates into a stair-step structure, with the stairs often formed by transverse roots of live trees. Steep slope thus provided a broader range of velocities and distinct habitat types (pool–riffle) in the forest streams, whereas the study reach in MT1 was an almost uniform glide.

Annual mean number of days with surface water over the study reaches during the 2-year recording period was: FT1 – 237 days; FT2 – 211 days; MT1 – 119 days; FE1 – 68 days; and FE2 – 84 days. In MT1, the wetted interval was considerably shorter during the 1989/90 season (101 days) compared to the previous year (139 days). In spite of an early spring drought causing partial desiccation in all channels, the mean number of days with surface water was almost constant for both years in the other streams.

Figure 2 illustrates drying patterns. The ephemeral forest streams dried out rapidly over most of the study reach but some seeps retained water for much longer. The temporary forest streams had ephemeral sections but also retained 1 or 2 summer pools or even small segments with seep-like trickles. About 5% of the overall channel length in FT1 and FT2 were permanently wetted. The meadow channels had no permanent pools, trickles or even seeps with moist or saturated soil. Instead, they dried out rapidly over the entire reach. No detailed records were obtained for the ephemeral meadow stream due to dense vegetation in the channel. Except for the shorter flow period, the drying pattern in ME1 closely resembled that of MT1.

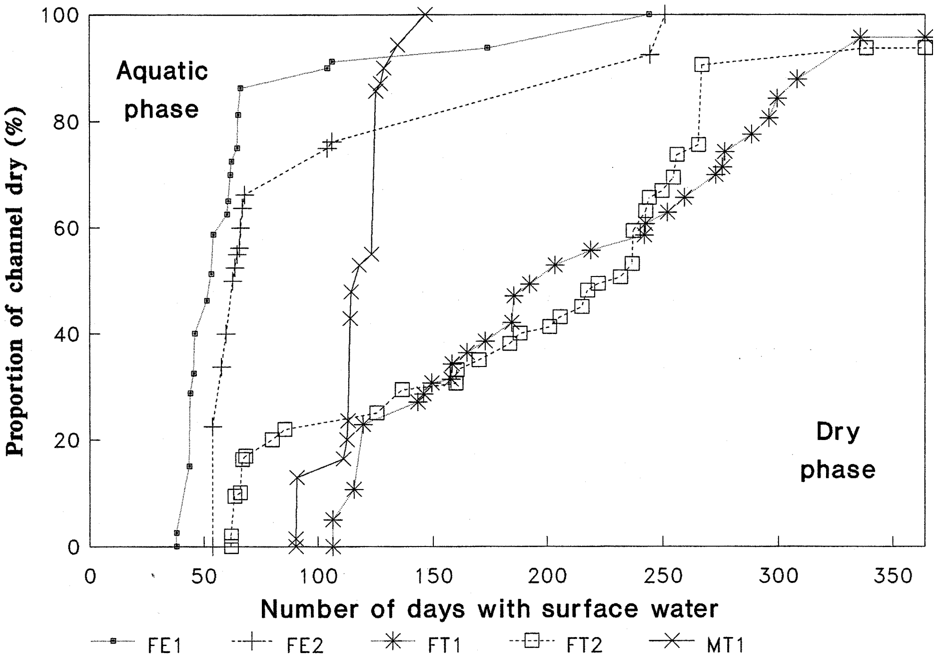

Figure 2. Drying patterns of 5 summer-dry streams. Drying patterns are expressed as proportion of the study reach without surface water after a given number of days (average over 2 seasons.)

Stream beds differed considerably with respect to sediment composition (Table 2). While the channel surface in FT2 and MT1 was largely dominated by rocky sediment, clay was the main component in FT1. Differences in stream-bed composition reflect the depth of the soils drained. A rocky stream bottom indicates comparatively shallow soils, whereas a clay dominated stream bed indicates deep soils.

Temperatures as high as 17.5 °C were recorded from the temporary meadow stream in mid-April, when flow had ceased in most of the study reach and the channel had begun to dry. Maximum temperature in the temporary forest streams was measured in July and August, 1990. Even then temperature in seemingly stagnant pools did not exceed 15 °C, in spite of air temperature commonly rising well above 30 °C. Seasonal fluctuations were less pronounced in both temporary forest streams than in the permanent channel (Figure 3).

Repeated measurements during the main flow period in winter and spring yielded oxygen concentrations at or above saturation in all streams. However, there was an O_2 gradient in the permanent FT1-pool (pool depth about 60 cm) in the autumn. O_2 in the litter accumulated during the stagnant phase was only 13% saturation; just above the litter layer O_2 was 41% saturation; and just below the water surface 60% saturation was recorded.

Sediment dynamics and retention

Sediment content in FE1 was highly variable and closely linked to intensity of road use and discharge. During the period of heavy road use (20–30 passes per day), suspended sediment levels were up to 100-fold higher in the section below the road than in the unimpacted reach. Short and intensive rainstorms a rapid increase in discharge and high amounts of sediment in the small channel. On January 9 suspended sediment increased from 30 mg l^{-1} to 2000 mg l^{-1} while discharge rose 10-fold within 4 h. Suspended sediment then decreased to 15 mg l^{-1} in the spot sample taken the following day (Figure 4).

At moderate concentrations (ca. 250–300 mg l^{-1}) in mid-February, 70–90% of the variation in ultra-fines (sediment < 1.6 μm) can be explained by distance from input (culvert) with a linear regression model. Content of ultra-fines decreased 9–26% over a 75 m reach. At low turbidity ($<$100 mg l^{-1}) no significant negative correlation was found between distance from input and the concentration of ultra-fines in the water. Rather, there was evidence for a positive correlation

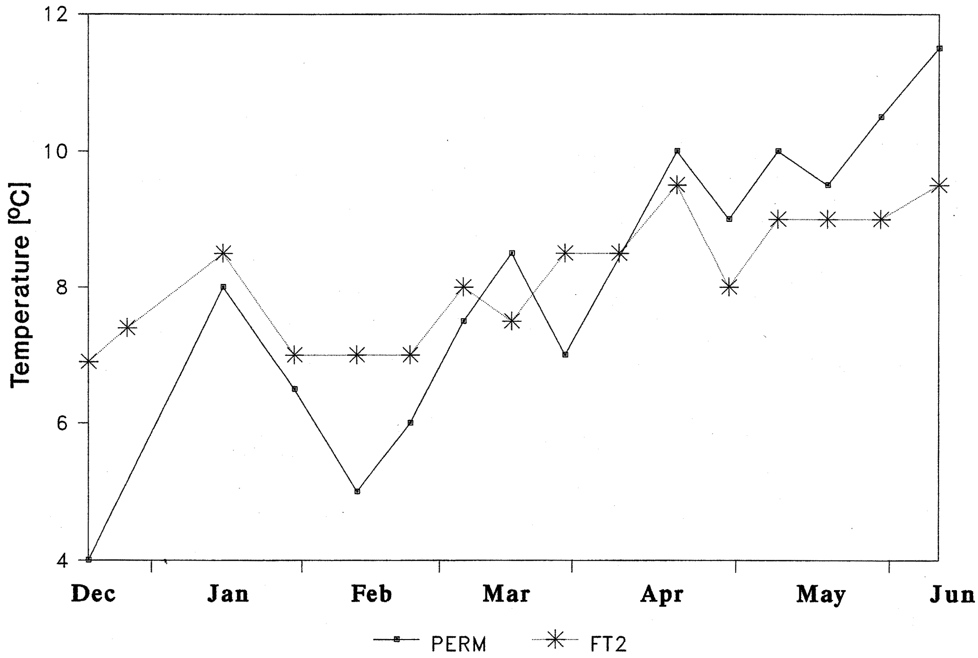

Figure 3. Comparison of water temperature in PERM and FT2. Measurements were taken at slightly irregular intervals between 18/12/89 and 15/07/90.

between distance from input and concentrations in FE1 (Table 3).

For suspended sediment (1.6 μm < suspended sediment < 53 μm) decline at moderate input levels (>150 mg l^{-1}) was 60–80% between sampling points at 25 and 100 m. Linear regression models relating suspended sediment in the water to distance from the road explained 70–80% of the variation in the data at low input levels (<50 mg l^{-1}) and at least 90% of the variation when concentrations at the culvert were moderately high (Table 3).

As the non-suspended sediment fraction (sediment > 53 μm) was quickly retained, correlation of the content in the water in relation to distance from input tended to be weaker than for ultra-fines and suspended sediment (Table 3).

The amount of suspended sediment trapped in leaf packs decreased with distance from input. At moderate concentrations in February, distance from the culvert accounted for 99% of the variation in the weight of suspended sediment trapped in FE3 packs ($p < 0.01$; $n = 5$). Over the same period only 35% of the variation was accounted for by distance from input in FE1 packs ($p < 0.05$; $n = 10$). However, negative correlations were tight for suspended sediment trapped in FE1 packs during the period of maximum input in January ($R^2 = 0.80$; $P < 0.01$; $n = 10$).

The comparison of core samples taken at a unique site of mayfly mass emergence in FE1 illustrates the negative impact of sediment deposition. Cores were obtained prior to the construction of the forest road in 1988 and again after the logging operation in 1990. The overall organic content in the cores decreased from 34% to 22% and core weight increased more than 4.5 times. Particle-size distribution was similar for both cores (Figure 5).

Emergence of *Paraleptophlebia gregalis* (Eaton) (Ephemeroptera, Leptophlebiidae) was ca. 10 000 m^{-2} in 1988, but only 1450 m^{-2} in 1990. Initially, the habitat for mayfly larvae on the pool bottom had a 3-dimensional character due to loose packing of particles. Mayfly larvae then moved freely within the substrate. Deposition of sediment from the road resulted in a much more densely packed substrate and, as a consequence, a major reduction in mayfly abundance.

Nitrate dynamics

Nitrate concentration in the temporary forest streams was highly variable on both a spatial and a temporal scale. Values in FT2 fluctuated between 26 and 817 μg l^{-1}. Nitrate content was consistently low ($\leq$15 μg l^{-1}) in both meadow streams (MT1 and ME1) and in FE1 (Table 4). Peak nitrate concentra-

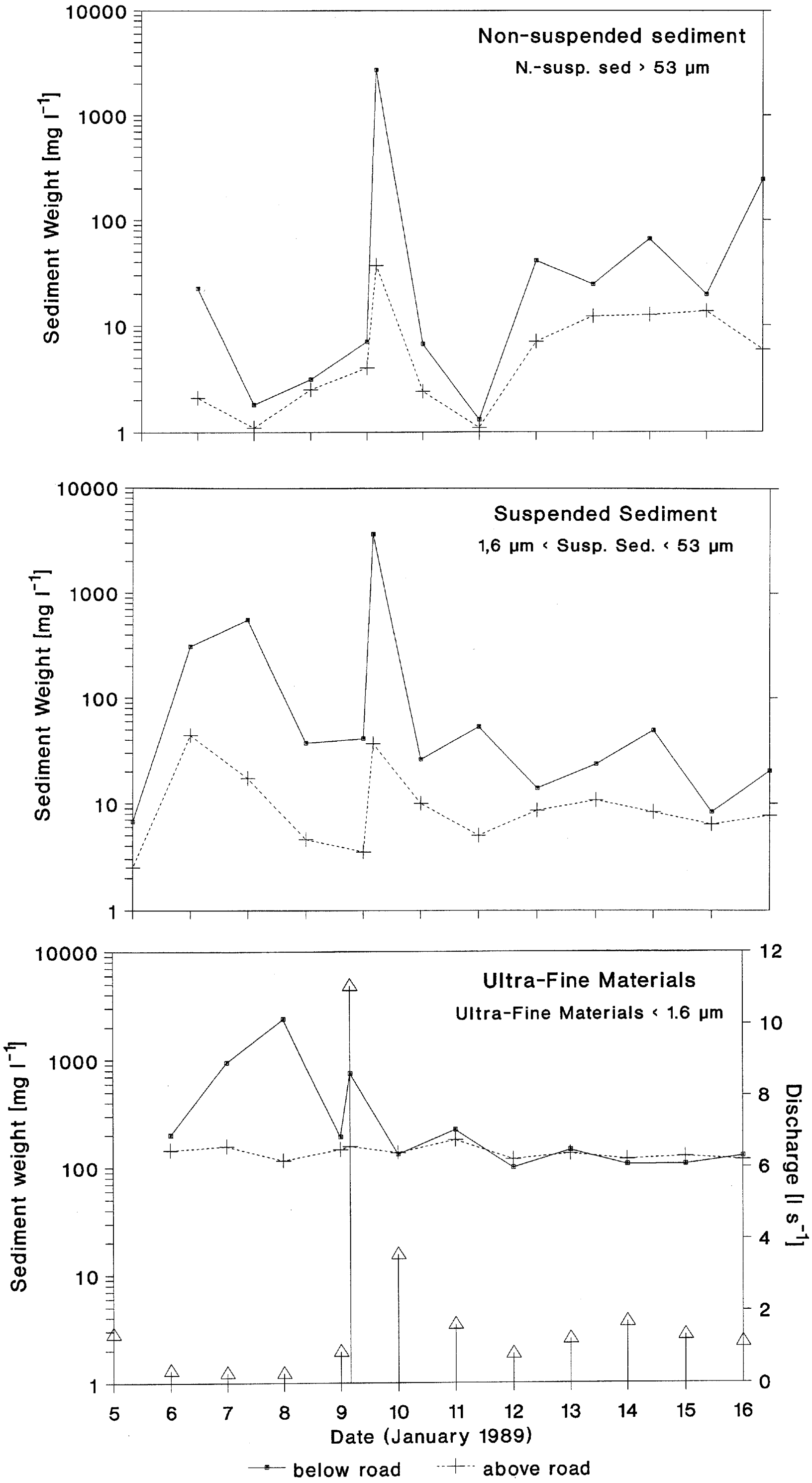

Figure 4. Sediment concentrations in the water (site FE1). Data points represent daily samples except that two samples were taken on 09/01/89 at 12:00 and 16:00. Bars in the lower diagram illustrate discharge at the culvert.

Table 3. Sediment removal from the water column. Linear regression of sediment concentration in the water as a function of distance from input

Date	Site	df	Non-suspended sediment			Suspended sediment			Ultra-fine materials		
			Slope	R^2	P	Slope	R^2	P	Slope	R^2	P
21/01/89[1]	FE1	2	+0.08	0.12	n.s.	−0.23	0.76	<0.07	+0.70	0.75	<0.08
	FE3	2	+0.53	0.73	<0.08	−0.10	0.78	<0.06	+0.12	0.17	n.s.
01/02/89[1]	FE1	2	−1.77	0.58	n.s.	−0.22	0.82	<0.05	+0.30	0.68	<0.10
	FE3	3	−1.31	0.94	<0.01	−0.37	0.70	<0.05	−0.09	0.08	n.s.
16/02/89[2]	FE1	2	−0.29	0.55	n.s.	−2.03	0.90	<0.05	−0.33	0.70	<0.09
	FE3	3	−0.38	0.53	<0.09	−0.99	0.98	<0.01	−0.60	0.91	<0.01
19/02/89[2]	FE1	2	−0.33	0.66	<0.10	−2.39	0.95	<0.01	−0.96	0.90	<0.05
	FE3	3	−0.15	0.58	<0.08	−1.87	0.95	<0.01	−0.71	0.83	<0.05

[1] Date with low sediment input levels.
[2] Date with elevated sediment input levels.

Table 4. N–NO_3 concentrations [μg l^{-1}] in the study streams. Day column indicates the number of days after flow resumed. Two locations separated by 114 m ($FT1_{12}$ and $FT1_{126}$) and 90 m ($FT2_{40}$ and $FT2_{130}$) were sampled in temporary forest streams. The abbreviation n.f. indicates no flow

Site	Preflow 02/11/90	2 days 26/11/90	20 days 13/12/89	90 days 23/02/90	125 days 27/03/90	190 days 19/06/90
$FT1_{12}$	15	12	8	10	3	8
$FT1_{126}$	256	166	82	61	27	64
$FT2_{40}$	817	354	152	59	65	174
$FT2_{130}$	26	239	100	58	37	79
FE1	n.f.	n.f.	12	15	16	n.f.
MT1	n.f.	n.f.	4	3	4	n.f.
ME1	n.f.	n.f.	8	9	n.f.	n.f.
PERM	4	9	52	59	31	47

tions were recorded before the onset of flow in both temporary forest streams. Pre-flow samples were taken from small pools that appeared in the channels when the water table had risen after several weeks with rain. Fluctuations in nitrate were less pronounced in PERM. The increase there lagged behind maximum values in FT1, its temporary headwater, and in FT2. Nitrate contents again increased in June after several days with rain (Table 4).

Nitrate concentration was not correlated with discharge during the first part of the flow period. It remained at 3 or 4 μg l^{-1} in MT1 while discharge varied between 0.1 and 3.0 l s^{-1}. In PERM and FT1, nitrate levels were almost identical for days 20 and 90, although discharge differed 10-fold in PERM and 20-fold in FT1. Nitrate content in FT2 did not differ between February 23 and March 27, whereas discharge ranged between 0.1 and 6.0 l s^{-1}.

During the second half of the flow period discharge was negatively correlated with nitrate in FT1, where three data points are available between days 125 and 190 (linear regression: R^2 = 0.99; $p < 0.05$). Similarly, nitrate in PERM increased as flow decreased between days 125 and 190. However, in FT2, which drains a shallow soil and flows on bedrock for some of its length, higher nitrate concentration in June 1990 coincided with an increase in discharge after a period of rain (Figure 6).

Solute injection experiments

Solute concentration, corrected for background, decreased 20–30% over 25 m in FT1 and 28–46% over 40 m in MT1. Passive tracer loss corresponded to exchange rates between stream water and subsurface flow of 0.8% (FT1, chloride) and 0.75% (MT1, rhodamine) per meter of channel. Uptake distance (S_W) was always less than 120 m for all solutes. The shortest overall uptake distance (64 m) was recorded for nitrate in MT1. Rhodamine WT in FT1 had the shortest solute turnover time (33 min) (Table 5).

In FT1 the N–NO_3/rhodamine WT and chloride/rhodamine WT ratios (corrected for background) at S_I were significantly smaller than those at S_D due to dye retention (t-test: $P < 0.01$; df = 15). In contrast, in MT1 nitrate was more efficiently removed from the water column than were the passive tracers (Table 5). The removal rate of nitrate was almost 18% above that of rhodamine WT. The N–NO_3/rhodamine WT ratio at S_I was significantly larger than that at S_D (t-test: $P < 0.01$; df = 14). There was an apparent decline of chloride exceeding that of rhodamine WT over the MT1 study reach. N–NO_3/chloride ratios at S_I and S_D

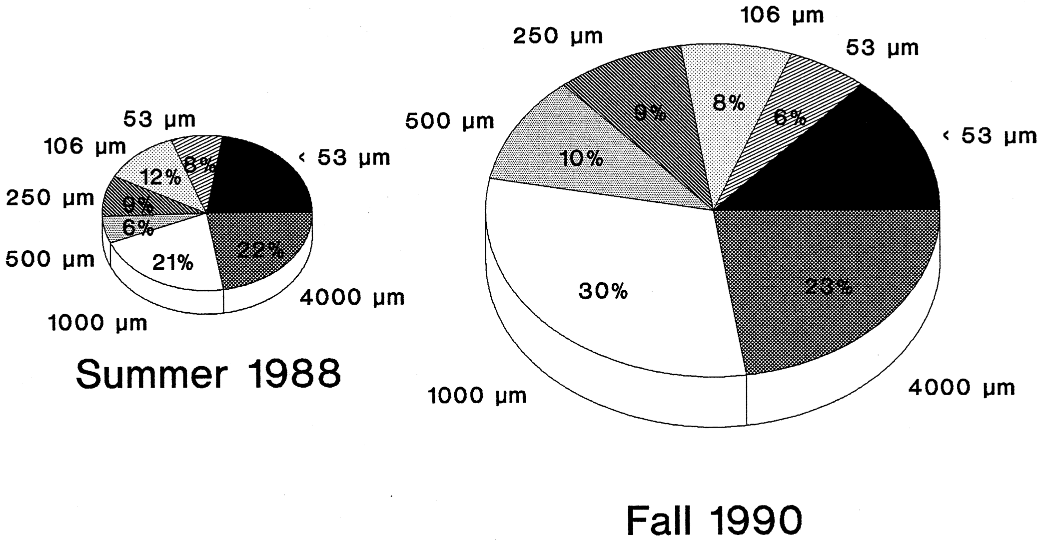

Figure 5. Effect of forest road sediment on stream bed structure. Core samples were taken from a sample site in FE1 before construction of the forest road (summer 1988) and after completion of the logging operations (autumn 1990). The size of the pies is proportional to the total weight/volume of the two cores.

did not differ significantly (t-test: P = n.s.; df = 14). However, due to variable chloride concentrations at the injection site, there was also no significant difference between S_I and S_D in the rhodamine WT/chloride ratios (t-test: $P < 0.1$; df = 14).

Estimated concentration in the subsurface solute pool differed from that in the stream (Table 6). While subsurface chloride concentration was always less than that in the streams, subsurface nitrate input raised the in-stream content of the nutrient in FT1 between S_I and S_D.

Discussion

Site characterization

Flow duration and summer-drought conditions are important variables determining habitat quality in temporary streams (Dieterich, 1992). The drying patterns recorded encompass the duration of the lotic and the transitional stagnant phase. There was a rapid transition from the lotic to the terrestrial stage in MT1, whereas the temporary forest streams dried slowly. The lotic phase in FT1 and FT2 was only about 4–6 weeks longer than in MT1, but the meadow stream lacked the stagnant phase with its seep and pool habitats. Presence of wetted microhabitats and moist soil are key factors determining survival of eggs and larvae during the summer drought and allowing colonization by larvae feeding on FPOM in water-saturated soil (e.g. craneflies) (Dieterich, 1992).

It is often postulated that temporary streams exhibit unusually high fluctuations in environmental variables (e.g. Boulton & Lake, 1990; Chapman & Kramer, 1991; Williams & Hynes, 1977). However, in the temporary forest streams we studied, environmental conditions during the lotic phase in many respects were more stable than in the permanent headwater. PERM had larger fluctuations in temperature than FT1 and FT2 because the latter were more closely connected to the groundwater and had fewer exposed rocks mediating heat transfer between air and water. In the temporary streams stair step structure dissipated water power. There was no large perturbation of bottom substrates and animal drift was extremely low even during peak floods (Dieterich, 1992). Thus, the degree of groundwater connection and the evaluation of morphological characteristics is needed to qualify statements about parameter variability in summer-dry streams. Colonization of these habitats is usually not limited by the range of physical and chemical variables characterizing the lotic phase, but rather by flow duration and channel characteristics during the summer drought.

Sediment retention

Ephemeral streams effectively retain sediment. The suspended sediment fraction was best suited to demon-

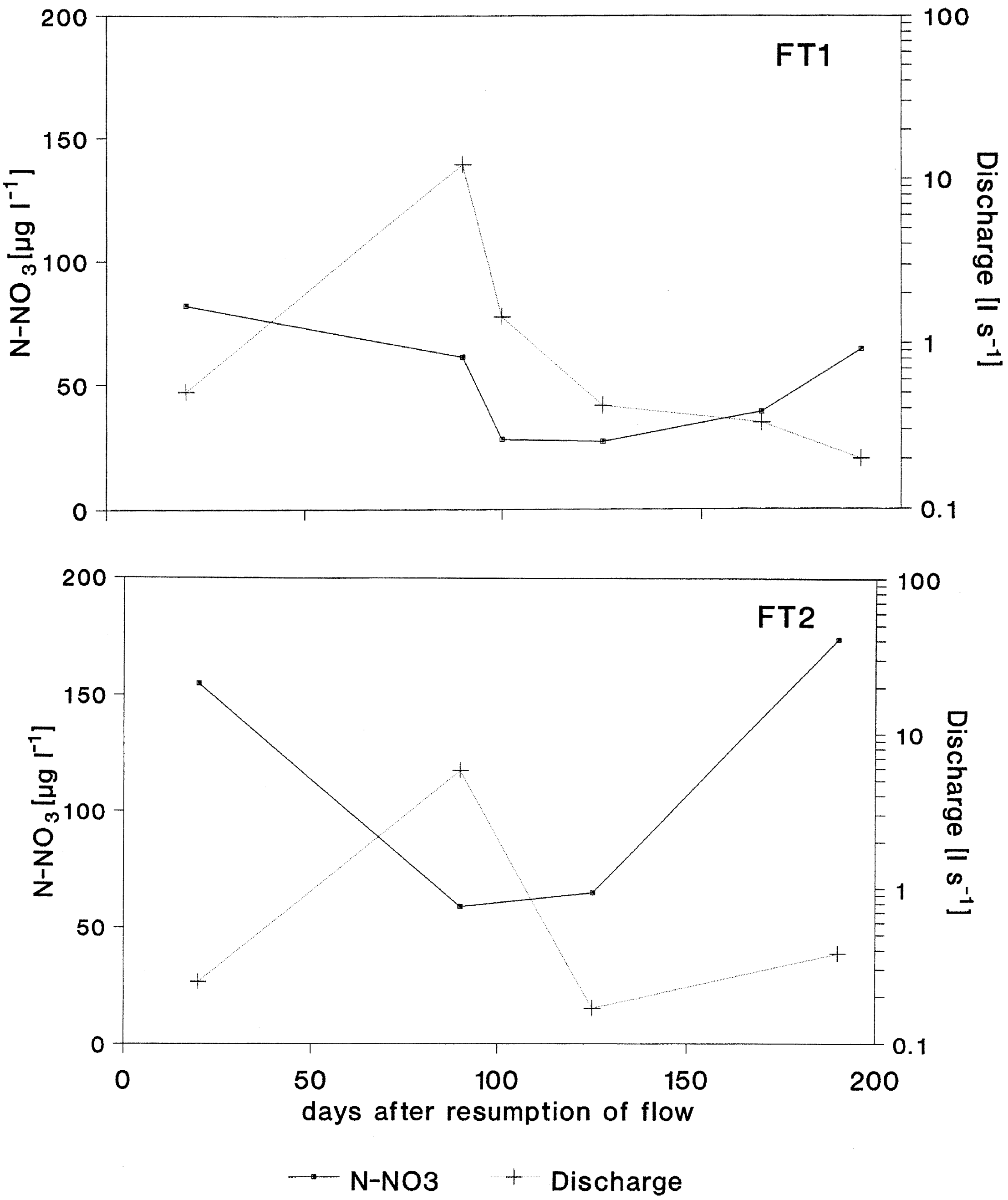

Figure 6. Discharge and nitrate patterns in temporary forest streams. Time period covered is from 13/12/89 to 19/06/90 (20–120 days after resumption of flow.)

Table 5. Summary of results from injection experiments. Two separate plateau-levels were established in FT1 (P_I and P_{II}). Solute plateau concentrations (±1 SE) are in mg l^{-1}. Data are corrected for background

Site	Solute	$Solute_I$	$Solute_D$	Δ Solute (%) (%)	S_w (m)	k_c^{-1} (min)
$FT1_{PI}$	Rhodamine	48.0 (±1.1)	33.1 (±1.3)	31.1	67	33
	Chloride	4.25 (±0.6)	3.42 (±0.3)	19.5	115	57
	Nitrate	0.132 (±0.003)	0.106 (±0.002)	19.7	114	57
$FT1_{PII}$	Rhodamine	74.4 (±4.9)	53.0 (±1.7)	28.8	74	–
	Chloride	6.8 (±0.2)	5.2 (±1.1)	23.5	93	–
	Nitrate	0.213 (±0.017)	0.170 (±0.005)	20.2	111	–
MT1	Rhodamine	120.9 (±1.6)	86.2 (±1.0)	28.7	118	78
	Chloride	57.3 (±8.8)	32.6 (±0.4)	43.1	71	47
	Nitrate	0.400 (±0.004)	0.215 (±0.009)	46.3	64	42

strate retention on the temporal and spatial scale used in this study. Non-suspended sediment was mostly deposited just beyond the culvert (spatial scale used too large). The ultra-fine materials were flushed out rapidly (time scale used too long). After the pool of ultra-fines on the road had been depleted, the addition of ditch water to the ephemeral streams caused a dilution of ultra-fine particles and solute concentrations in the stream water. Hence, the recovery of FE1 to its 'natural' state translated into an enrichment of the water with ultra-fine materials from the forest soils.

Fine particles accumulate in pools because flow conditions there are not favorable for them to be swept into suspension again (Graham, 1990). But resuspension relative to deposition becomes increasingly less important as sediment concentration in the water increases. The noise in the leaf pack data caused by different rates of sediment resuspension between packs exposed in riffles and those in pools causes less tight correlations between distance from input and weight of sediment trapped if particle concentrations are low. This is reflected in the lower R^2 values for samples taken in February (sediment input moderate) as compared to those taken in January (sediment input high). It is also reflected in the higher variation in the weight data for packs exposed more distant from the culvert (FE1 packs).

The filtration efficiency of ephemeral headwater streams results from the shallowness of the water column combined with the large number of retentive structures – both organic debris and living plants (grass, mosses) – and, more importantly, from the pattern of lateral and longitudinal expansion and contraction in response to precipitation. In small headwaters, spates may peak shortly after the heaviest rain and last only a few minutes before receding (Leopold, 1994). During expansion, water movement at the 'trickle-front' is mostly into the soil. Therefore, much of the sediment in the stream water is actually filtered through the soil rather than being flushed downstream. Filtration efficiency is expected to increase as flow periods become shorter after cessation of rain. Short flow periods translate into frequent channel rewetting and thus a higher potential for maximum filtration efficiency following precipitation.

Table 6. Solute concentrations (mg l^{-1}) in the subsurface pool (Subpool) over the injection reach. Estimation of groundwater exchange are based on chloride loss (FT1) and rhodamine loss (MT1)

Solute	FT1			MT1		
	S_I	S_D	Subpool	S_I	S_D	Subpool
Chloride	4.61	4.25	3.4	3.9	3.26	1.8
Nitrate	0.038	0.043	0.055	0.005	0.004	0.002

Bilby & Likens (1979) found that during a flood, seston peaks occurred before the maximum discharge. For a small first-order stream Naiman (1982) recorded peak sediment concentrations preceding the rise in discharge. This concentration–discharge pattern increases the overall efficiency of a filtration process coupled to lateral and longitudinal expansion. If sediment concentration is highest during the early flood stage, then filtration efficacy and turbidity are at a maximum simultaneously. Finally, the rapid recession of stream width after a spate minimizes resuspension and adds further to the retention capacity of the system.

Field observations after the first year of road use suggest that fine sediment deposited in a summer-dry stream may be incorporated into larger, non-suspendable soil particles during the dry season. Thus, rather than being temporarily stored, fine sediment can be removed permanently from the aquatic environment. This amounts to a seasonal regeneration of the fine sediment filtration and storage capacity of summer-dry streams.

Nitrate dynamics

The physical and chemical features of low order streams closely reflect small-scale differences in soil, geomorphology and vegetation. This is evident in the nitrate patterns observed in FT1 and FT2 on a spatial and on a temporal scale. Recently, Maltchik et al. (1996) also reported a strong spatio-temporal heterogeneity of nutrient concentration from a Mediterranean temporary stream.

Temporal heterogeneity results from flushing of solutes accumulated in the soil during the dry period (Everard, 1996; Pinol et al., 1992). Nitrate loss increases in soils exposed to cycles of drying and wetting (Patrick & Wyatt, 1964; Reddy & Patrick 1975). Drying creates aerobic conditions that promote nitrification, whereas denitrification is the dominant transformation process in soils at $>$60% of their water-holding capacity (Alexander, 1977). In McDonald forest, nitrate formed under aerobic conditions in summer was flushed from the soils and appeared in the streams immediately before and just after resumption of flow. Flushing depletes the nitrate pool and denitrification in the saturated soil will further limit nitrate export from the terrestrial to the lotic system. This is reflected in lower in-stream nitrate concentrations later in the season.

Nitrate concentration in FT1 and PERM increased when discharge decreased in the second half of the flowing season. Triska et al. (1990) attribute the increase of in-stream nitrate in summer to a proportional increase in the amount of water with long residence time in the soil. The groundwater is nitrate rich, whereas the contribution of nitrate-depleted water from shallow subsurface flow decreases as the dry season progresses. However, in FT2 an increase in nitrate paralleled by an increase in discharge was observed in June after a period of rain in late spring. Just as early in the flow period, this increase may reflect nitrate flushing from the soil. Thus, depending on geomorphological conditions, both flushing and the relative contribution of groundwater to stream flow are important determinants of in-stream nitrate concentrations in summer-dry headwaters.

Spatial nitrate patterns within the temporary forest streams may primarily be caused by water routing through the soil and subsequent mixing of water from different subsurface sources. Hornberger et al. (1991) showed that transport of water in forest soils is controlled by 'preferred paths'. The contraction and expansion of these paths as regulated by precipitation will determine the overall efficiency of nitrate flushing as well as the extent of the water-saturated, anaerobic zone in the soil. For springs, mixing of groundwater flow is a key parameter determining water quality (van der Kamp, 1995). The temporary forest streams, characterized by summer seeps, pools and trickles may be viewed as channels linking different sites of groundwater discharge. Groundwater qualities therefore will be closely reflected in the surface flow.

The temporary meadow stream and the ephemeral channels lack a tight connection to the groundwater. Triska et al. (1990) found nitrate concentration in a headwater stream in northwest California to be low when the soil was saturated and rain was 'quickly converted to discharge by rapid, shallow subsurface flow'. Weekly N–NO_3 concentration in rainwater at a weather station near Corvallis ranged between 21 and 83 μg l^{-1} (average 54 μg l^{-1}) from January to April 1989 (NADP, 1990). Denitrification and uptake by the terrestrial community are believed to reduce the nitrate content in the rainwater as it moves towards the meadow stream and the ephemeral channels as shallow subsurface flow.

Injection experiments

Uptake distances over the injection reach in FT1 and MT1 were generally less than 120 m for all solutes. This is within the range in headwater streams reported elsewhere for phosphate (Mulholland et al., 1985; Newbold et al., 1983) and nitrate (Triska et al., 1990). Solute loss in both streams is attributed mainly to groundwater exchange rather than to biological uptake. Rhodamine WT loss exceeding that of chloride and nitrate in FT1 indicates that the dye is not a passive tracer there. Sorption to sediment and organic matter was found to reduce rhodamine WT concentration in the water column of headwater streams (Munn & Meyer, 1988; Smart & Laidlaw, 1977). Such interactions of rhodamine WT with the sediment are poorly understood and depend on site-specific conditions (Bencala et. al., 1983).

Chloride is considered a more reliable passive tracer and thus often preferred over rhodamine WT in solute injection experiments. The possibility of chloride retention in the meadow stream warrants caution in the use of chloride as the single passive tracer in pulse injection experiments. Further studies are required to determine if chloride can be retained in nitrate-limited systems characterized by low flow, a neutral pH and dense aggregations of filamentous algae as stream width recedes late in the season.

Newbold et al. (1982) argue that the influence of biological processes on the concentration of a given nutrient increases as that nutrient becomes more limiting to system productivity. Nitrate limitation is considered to be a specific feature of streams in the Pacific Northwest (Munn & Meyer, 1990). Low in-stream concentrations (3–4 μg l^{-1}) and the absence of a diel fluctuation in N–NO_3 content provide evidence that MT1 is nitrate limited at all times (Dieterich, 1992). Biological retention of nitrate is important there, as significantly more nitrate than rhodamine WT was removed over the injection reach. In contrast, there was no indication of biological uptake in the forest streams. This agrees with findings by Gregory (1980) and Lowe et al. (1986) that primary production is light-limited in forested headwaters and, therefore, insensitive to nutrient enrichment.

Conclusions

Summer-dry streams have a high potential to modify habitat conditions in permanent lotic systems. This study shows that ephemeral channels are highly efficient at filtering sediment from the water column. The fertilization experiments demonstrate the potential for nutrient/pollutant removal in summer-dry streams. The removal of nutrients such as nitrate is greatly enhanced through large boundary surfaces and long solute residence times.

A distinct difference remains between the removal of sediment and the removal of solutes such as nitrate. In contrast to sediment there is a potential for solutes to be transferred from the surface to the subsurface flow. The question of whether nitrate lost to subsurface flow will end up in groundwater reservoirs, terrestrial plants, microflora, or in the atmosphere (denitrification) is important if the role of summer-dry streams in the landscape is to be assessed. If temporary or ephemeral streams just help to channel solutes from surface water to groundwater systems, then pollution problems have not been solved, but just been shifted elsewhere. More research in this field is needed. Because of the small catchment area they drain and the visible adherence to underground flow, summer-dry headwaters are particularly well suited to study both the interdependency of terrestrial and aquatic ecosystems and the connection between surface run-off and subsurface flow including groundwater reservoirs.

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