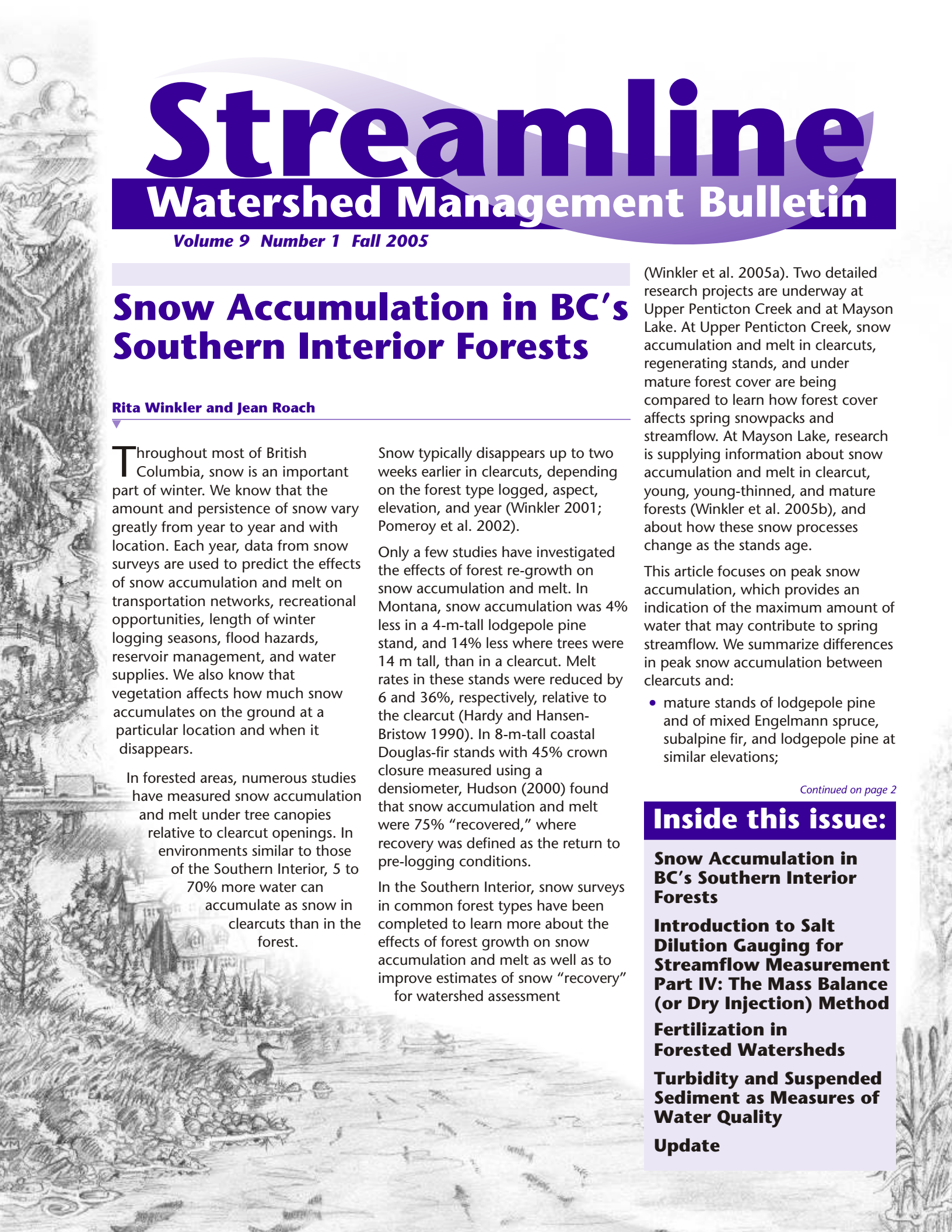




Streamline

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Snow Accumulation in BC's Southern Interior Forests

Rita Winkler and Jean Roach

Throughout most of British Columbia, snow is an important part of winter. We know that the amount and persistence of snow vary greatly from year to year and with location. Each year, data from snow surveys are used to predict the effects of snow accumulation and melt on transportation networks, recreational opportunities, length of winter logging seasons, flood hazards, reservoir management, and water supplies. We also know that vegetation affects how much snow accumulates on the ground at a particular location and when it disappears.

In forested areas, numerous studies have measured snow accumulation and melt under tree canopies relative to clearcut openings. In environments similar to those of the Southern Interior, 5 to 70% more water can accumulate as snow in clearcuts than in the forest.

Snow typically disappears up to two weeks earlier in clearcuts, depending on the forest type logged, aspect, elevation, and year (Winkler 2001; Pomeroy et al. 2002).

Only a few studies have investigated the effects of forest re-growth on snow accumulation and melt. In Montana, snow accumulation was 4% less in a 4-m-tall lodgepole pine stand, and 14% less where trees were 14 m tall, than in a clearcut. Melt rates in these stands were reduced by 6 and 36%, respectively, relative to the clearcut (Hardy and Hansen-Bristow 1990). In 8-m-tall coastal Douglas-fir stands with 45% crown closure measured using a densiometer, Hudson (2000) found that snow accumulation and melt were 75% "recovered," where recovery was defined as the return to pre-logging conditions.

In the Southern Interior, snow surveys in common forest types have been completed to learn more about the effects of forest growth on snow accumulation and melt as well as to improve estimates of snow "recovery" for watershed assessment

(Winkler et al. 2005a). Two detailed research projects are underway at Upper Penticton Creek and at Mayson Lake. At Upper Penticton Creek, snow accumulation and melt in clearcuts, regenerating stands, and under mature forest cover are being compared to learn how forest cover affects spring snowpacks and streamflow. At Mayson Lake, research is supplying information about snow accumulation and melt in clearcut, young, young-thinned, and mature forests (Winkler et al. 2005b), and about how these snow processes change as the stands age.

This article focuses on peak snow accumulation, which provides an indication of the maximum amount of water that may contribute to spring streamflow. We summarize differences in peak snow accumulation between clearcuts and:

- mature stands of lodgepole pine and of mixed Engelmann spruce, subalpine fir, and lodgepole pine at similar elevations;

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Suite 702, 235 1st Avenue
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Project Manager:

Robin Pike
Tel: (250) 387-5887

Distribution/Mailing List:

Janet Jeffery
Tel: (250) 371-3923

Technical review committee:

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Cover Illustration: **William McAusland**
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- two mature pine stands at different elevations; and
- young pine and mature mixed-species stands.

The results of the stand comparisons presented in this article are the background to ongoing research at both Upper Penticton Creek and Mayson Lake.

Study Areas

The Mayson Lake snow research area is located 50 km northwest of Kamloops, BC (Figure 1), in the dry Montane Spruce biogeoclimatic zone (Lloyd et al. 1990). This area receives about 700 mm of precipitation annually, of which about 60% falls as snow. From 1995 to 1997, four adjacent and relatively flat sites at an elevation of 1250 m were surveyed. One site was forested with multi-layered mature Engelmann spruce, subalpine fir, and lodgepole pine. Two sites were forested with 4- to 6-m-tall young lodgepole pine, one of which was thinned and pruned. A fourth site was a reforested clearcut with young lodgepole pine trees less than 1 m tall. These same stands are now being resurveyed as part of an ongoing study into changes in snow accumulation and melt associated with forest growth.

The Upper Penticton Creek study area is located 26 km northeast of Penticton, BC (Figure 1), in the dry Engelmann Spruce–Subalpine Fir biogeoclimatic zone (Lloyd et al. 1990). Annual precipitation in this area is about 750 mm, of which approximately 50% falls as snow. From 1995 to 1997, snow surveys were completed at five sites: a mature lodgepole pine stand; a mature mixed Engelmann spruce, subalpine fir, and lodgepole pine stand; clearcuts with no young trees taller than 1 m located next to each of the mature forest sites; and a 4-m-tall stand of young mixed spruce, fir, and pine. These sites are all at an elevation of 1600 to 1700 m on level terrain. The mature lodgepole pine stand and adjacent clearcut continue to be surveyed and now provide 10 years of data.

Beginning in 1998, a second mature lodgepole pine stand, with lower stocking density, was also included in the study at Upper Penticton Creek. This stand at 1650 m is located between the mature lodgepole pine stand and nearby clearcut. In 2000, two additional snow survey sites, a recently logged clearcut, and an adjacent mature high-elevation lodgepole pine stand were established at 1930 m. Five years of data have been collected at these sites.

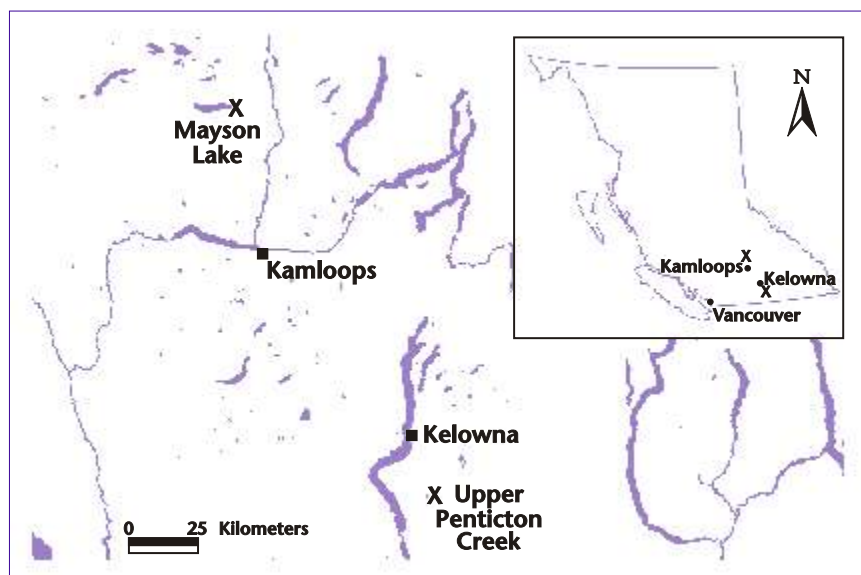


Figure 1. Location of the Mayson Lake and Upper Penticton Creek snow research sites.

Detailed forest inventories have been completed for every study site. Variables measured included stand density, crown closure measured with a moosehorn (Bunnell and Vales 1989), stand basal area, average tree diameter, average tree height by canopy layer, and average crown length. Using these data, Winkler et al. (2005a) found that crown closure measured with a moosehorn best explained the largest proportion of the variability (> 68%) in snow accumulation among the study stands.

The response of SWE to forest cover, including regeneration, is not a simple matter — aspect, species mix, and elevation all play a role.

Snow Survey Methods

The amount of water in the snowpack, or snow water equivalent (SWE), was measured at all sites on or near April 1 using a standard Federal snow tube. At each site, snow survey sample points were spaced in a 15 × 15 m or 10 × 10 m grid pattern over approximately 1 ha. From 1995 to 1997, 64 sample points were surveyed in each stand. From 1998 on, snow has been surveyed at 32 points within each stand, a more manageable sample size that is still large enough to reflect the variability in snow accumulation expected in the forest types being studied (Spittlehouse and Winkler 1996).

Results

The SWE measured at Mayson Lake and Upper Penticton Creek on or near April 1 from 1995 to 2004, along with the attributes of each stand surveyed, are shown in Table 1. The percentage reduction in SWE under each forest cover type relative to that measured in the adjacent clearcut is also provided.

Our results show some interesting differences in April 1 snow accumulation between stands. Average differences were smaller between the mature pine stands and adjacent clearcut (10% and 22%, respectively, at Upper Penticton

Creek) than between the mixed spruce stands and adjacent clearcuts (23% and 32% less SWE at Upper Penticton Creek and at Mayson Lake, respectively). At Upper Penticton Creek, a young 4-m-tall mixed

spruce–fir–pine stand with minimal crown closure accumulated slightly more snow (3%) than an adjacent clearcut. The increase in accumulation in the young stand may simply have been due to location or may have been a result of reduced sublimation losses. Although the increase was consistent from year to year it is also within measurement

error. In the 4- to 6-m-tall pine stands at Mayson Lake, SWE was 14% less than in the clearcut indicating a 43% recovery in snow accumulation at a canopy closure of 20 to 30%. On average, no difference in April 1 SWE was observed between the thinned and unthinned young pine stands at Mayson Lake, likely a result of the similarities in crown closure (21% and 28%, respectively) even though stem density had been reduced in the thinned stand. The difference

between stands varies from year to year and the range in SWE measured at each site is often larger than the differences among sites, highlighting the importance of sampling over a number of years.

Elevation

We also looked at the effect of elevation on snow accumulation. Unexpectedly, the April 1 SWE was 22% higher in the upper-elevation pine stand than in the adjacent clearcut. This finding is thought to be a result of increased snow sublimation losses and intermittent late-winter snowmelt in the southerly oriented high-elevation exposed clearcut. SWE in the high-elevation mature pine stand was 37% greater than in the lower-elevation pine stand, whereas the upper-elevation clearcut (1930 m) showed 13% less SWE than the lower-elevation SWE site (1650 m) on April 1. These results illustrate the spatial complexity of snow processes and are important in understanding how melting snow in different parts of a watershed contributes to streamflow.

Crown Closure

We also looked at the potential relationship between crown closure and the reduction in SWE measured

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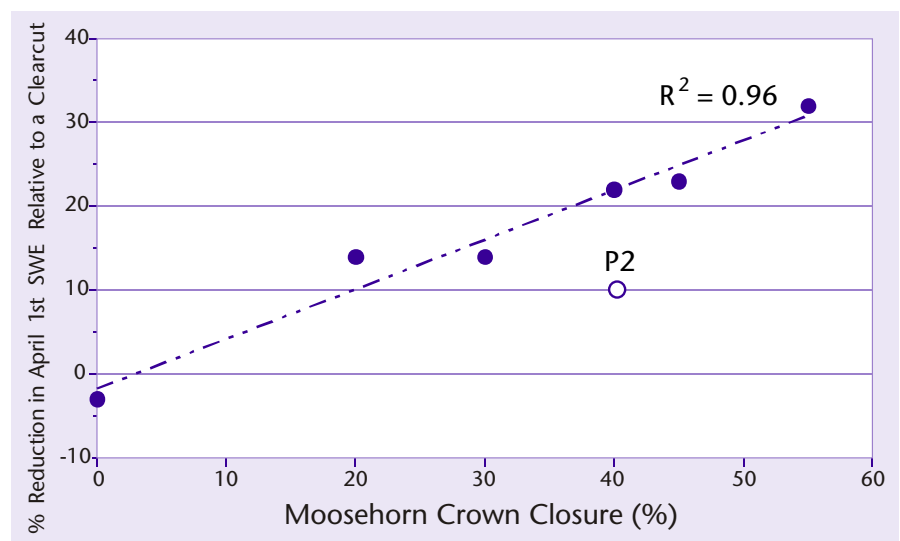


Figure 2. Reduction (%) in peak snow water equivalent (SWE), measured on or near April 1, under forest cover relative to a clearcut, in seven stands at Mayson Lake and Upper Penticton Creek.

Table 1. Stand characteristics and percentage reduction in snow water equivalent (SWE) measured on or near April 1 at Mayson Lake and Upper Pentiction Creek

Location	Site name	Stand type	Elevation (m)	Survey years	Sample sites	Main canopy tree height (m)	Basal area (m ² /ha)	Crown closure (%)	April 1 SWE (cm)		Reduction in SWE (%)
									Average	Range	
Mayson Lake	M1	Clearcut	1250	1995–1997	64	<1		0	22	20–26	
	M2	Young-thinned pine	1250	1995–1997	64	6	8	21	19	17–24	14
	M3	Young pine	1250	1995–1997	64	4	12	28	19	15–23	14
	M4	Mature spruce, fir (pine)	1250	1995–1997	64	23	62	55	15	12–19	32
Pentiction Creek	P1	Clearcut	1650	1995–2004	(64*)32	<1		0	31	23–42	
	P2	Mature pine	1670	1995–2004	(64*)32	18	58	40	28	17–37	10
	P7	Mature pine	1650	1995–2004	32	24	38	39	24	15–34	22
	P3	Clearcut	1600	1995–1997	64	<1		0	31	28–35	
	P4	Mature spruce, fir (pine)	1600	1995–1997	64	19	48	44	24	22–26	23
	P5	Young spruce, fir (pine)	1600	1995–1997	64	4	6	<10	32	29–37	–3
	P9	Clearcut	1885	2000–2004	32	<1		0	27	20–32	
	P10	Mature pine	1920	2000–2004	32	13	42	42	33	22–43	–22

*64 points were sampled each year from 1995 to 1997, subsequently 32 of these original points were sampled each year from 1998 to 2004.

Continued from page 3

on or near April 1. Among most of the study sites, we found a strong relationship between the two variables (Figure 2). The exceptions were the upper-elevation pine stand (for reasons already discussed) and the lower-elevation pine stand P7. Stand P7 had similar crown closure to the second nearby pine stand (P2), but the length of live crowns was much shorter in P2 than P7. The difference in canopy structure resulted in only a 10% reduction in SWE from that in the clearcut at P2 compared with a 22% reduction at P7. These exceptions clearly illustrate the complex interactions between forest cover and snow accumulation. For most of the stands included in this study, our results indicate that on average about a 6% reduction in SWE may be expected for roughly every 10% increase in crown closure. This relationship will be tested across a broader range of stand types but is similar to the findings of Moore and

McCaughey (1997). This type of information would give a useful index to changes in snow accumulation as a result of logging and forest re-growth for watershed planning.

Operational Interpretations

- Average differences in snow accumulation from one year to the next can be greater than the difference between peak SWE in forests and openings in the same year.
- Clearcuts in mixed spruce–fir–pine stands may result in larger increases in snow accumulation than clearcuts in pine stands.
- Though snow accumulation generally increases with elevation in a watershed, other variables such as aspect and exposure may have a greater effect on peak SWE.
- More snow may accumulate in young forests than in a clearcut

during early stages of stand development.

- Measurable reductions in snow accumulation occur as crown closure increases, as shown by the 6% reduction in SWE with every 10% increase in crown closure in most of the stands studied.
- The results presented here illustrate that the response of SWE to forest cover, including regeneration, is not a simple matter — aspect, species mix, and elevation all play a role.

Continuing Work

Snow research in the Southern Interior of British Columbia is continuing to answer questions related to forest cover effects on snow accumulation and melt. The first set of snow accumulation and melt recovery surveys at Mayson Lake were completed in spring 2005 and these stands will be resurveyed beginning in



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2008 to learn more about the effects of forest growth on snow accumulation and melt. Ongoing work at Upper Penticton Creek will offer information linking snow processes to spring streamflow. New work in burned stands and areas affected by mountain pine beetle will add to our understanding of snow accumulation and melt in the Southern Interior.

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For further information, contact:

Rita Winkler

Forest Hydrologist
BC Ministry of Forests and Range
Southern Interior Region
Kamloops, BC
Tel: (250) 828-4162
E-mail: Rita.Winkler@gems7.gov.bc.ca

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Introduction to Salt Dilution Gauging for Streamflow Measurement Part IV:

The Mass Balance (or Dry Injection) Method

Rob Hudson and John Fraser

In part one of this series, Moore (2004a) introduced the general principles of stream gauging by salt dilution. In subsequent articles, Moore (2004b, 2005) described techniques of constant-rate injection and slug injection using salt in solution. This is the final article in the series and details the "mass balance method." Originally described by Elder et al. (1990), the mass balance method differs from slug injection using salt in solution (Moore 2005) in that it is based on conservation of tracer mass, not of tracer volume. By using the mass balance method, salt can be injected into the stream either in dry form or in solution. Because it is more common to work with dry salt, this method has also become known as the "dry injection method."

Dilution methods have been used for at least five decades (Østrem 1964; Church and Kellerhals 1970; Church 1975; Day 1976, 1977a, 1997b; Hongve 1987; Johnstone 1988; Kite 1993). In 1998 we began to develop the mass balance method for use in small BC streams where current metering is

either difficult or impossible (Hudson and Fraser 2002). Subsequently, we have applied this method successfully in several coastal streams including Russell Creek (Hudson and Fraser 2002), upper Nahmint River on Vancouver Island, Flume Creek (Sunshine Coast), and Culliton and Furry Creeks (Sea-to-Sky Highway).

Background

The basic principle of dilution gauging is to add a known quantity of a tracer to a stream and observe its concentration in the stream at a point where it is fully mixed with the flow. The higher the flow, the more it dilutes the tracer. Dry salt used as the tracer must be injected at a point that favours rapid dissolution. This creates a salt solution *in situ* that then disperses into the flow aided by turbulence in the water column. The resulting concentration of salt is measured as electrical conductivity at a point downstream of the injection point where it is completely mixed. The distance

between the injection and measurement points is known as the mixing length (L). The dispersion pattern of conductivity over time is similar in shape to a storm hydrograph (Figure 1).

The mass balance method is based on conservation of tracer mass, not of tracer volume.

Streamflow Q is calculated using equation 1 where M is the mass of salt (in grams) and A is the area under the graph of concentration over time (Figure 1). The units of A are milligram-seconds per litre (equivalent to $\text{g} \cdot \text{s}/\text{m}^3$).

$$Q = \frac{M}{A} \quad (1)$$

The quantity A in equation 1 and shown on the graph (Figure 1) can be calculated as:

$$A = \int c_t \, dt \quad (2)$$

where c_t is the concentration of injected salt at time t , and t_{int} is the time interval between successive data points.

As noted above, the salt concentration is measured as electrical conductivity (EC) in the stream. The concentration of the injected salt can be calculated using equation 3 below:

$$c_t = (EC_t - EC_0) \cdot CF \quad (3)$$

where EC_t is the electrical conductivity at time t , EC_0 is the baseline conductivity, and CF is the concentration factor. The concentration factor is the coefficient in the near-linear relation between EC and salt concentration. However, CF is not a constant, since stream temperature and background chemistry also affect EC. These effects constitute a potential source of error that can only be controlled by understanding the relationships between EC and salt concentration, stream temperature, and chemistry.

Factors Affecting EC versus Salt Concentration¹

For a salt dilution measurement, both the background ion concentration in the stream and the concentration of salt added to the stream affect the CF . Hongve (1987) found that the

¹ Within the salt dilution gauging literature, there is a lack of agreement about the derivation of a unified calibration approach. We have developed the procedures presented in this article as one example of an approach that could be used and (or) modified by the reader. These procedures are based on trials that we conducted for equipment, streamflows, and watersheds we commonly sampled. As with any measurement technique, procedural steps and calibration assumptions should be validated for the local watersheds in which the technique is applied.

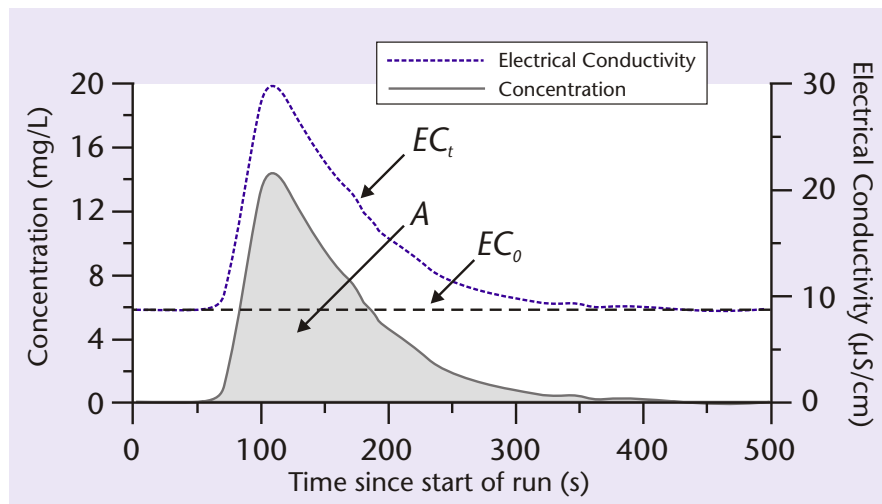


Figure 1. Salt dilution measurement at Russell Creek, October 2003. The shaded area is the quantity A that must be calculated.

concentration factor was directly related to the baseline conductivity of the stream. For a pure NaCl solution, 1 mg of NaCl added to 1 L of de-ionized water will increase conductivity by 2.14 $\mu\text{S}/\text{cm}$ ($\text{CF} = 0.467$). As the ionic strength of the solution increases, the interaction of ions in solution begins to hinder each others' activity. The CF is equal to 0.47 for [NaCl] in the range from 0 to 30 mg/L (EC from 0 to 64 $\mu\text{S}/\text{cm}$) and increases to 0.51 for [NaCl] in the range from 300 to 1000 mg/L (EC from 617 to 1990 $\mu\text{S}/\text{cm}$). However, the presence of ions other than NaCl in the stream being measured may result in a slightly different range of values for the CF.

The relationship between EC and temperature is more-or-less linear in the range of temperatures commonly encountered during flow measurements, but there is considerable lack of agreement in the literature concerning that relationship at low temperatures (i.e., 0–3°C). Smart (1992) found a linear relation for temperatures ranging from less than 1 to 10°C, contradicting statements by Østrem (1964) and Collins (1978). Johnstone (1988) reported linear relations for temperatures from 0.5 to 25°C.

We have spent considerable effort in trying to calibrate our conductivity probes and believed we had a reasonable temperature correction curve based on a combination of field and laboratory calibrations (Figure 2). However, it has now become evident that differences in water chemistry among different streams have a significant effect on the value of CF such that partial calibrations from different sites cannot be combined. Rather, a separate temperature correction curve must be developed for each stream, and laboratory calibrations are not necessarily applicable in the field. We have therefore used a simple calibration procedure to derive a specific value of CF for each measurement. A simple calibration kit and procedure are described below.

Probe Calibration²

Calibration kit

The calibration kit should consist of the following items:

1. A 1-L graduated cylinder
2. A 10-mL glass pipette and filler bulb
3. Standard solutions of 2000 and 20 000 mg/L NaCl
4. Distilled or de-ionized water

Calibration procedure

This procedure should be performed each time a measurement is collected.

1. Allow the EC probe to equilibrate to stream temperature. Note the EC and the temperature.
2. Obtain a 980-mL sample of the streamwater in the graduated cylinder.
3. Place the probe in the cylinder and note the EC and temperature. These values should be close or identical to the values obtained in step 1. If so, they represent baseline values. If not, get a different sample.

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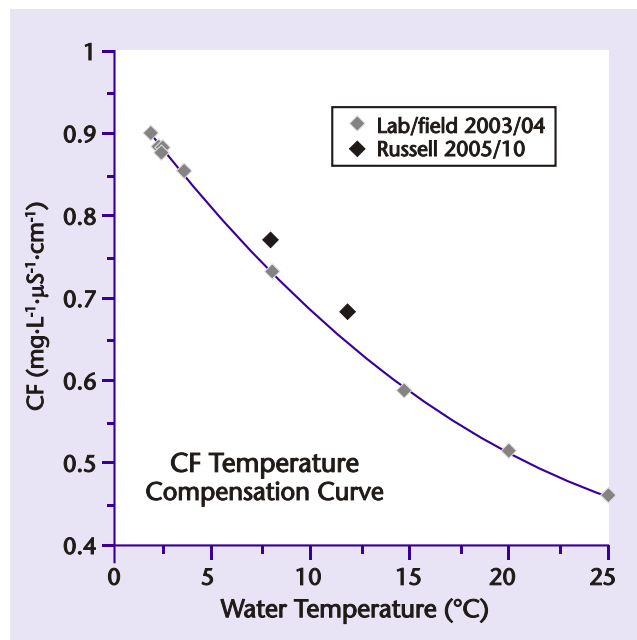


Figure 2. This CF temperature curve was developed in the lab and at a field site (Furry Creek) where baseline EC was very low (about 10 $\mu\text{S}/\text{cm}$). However, as noted by Hongve (1987), the stream chemistry affects the value of CF such that at Russell Creek, where baseline EC was higher, the CF is also slightly higher.

² The procedures presented illustrate one approach to probe calibration; other approaches may be used or need to be developed in the application of the mass balance method to meet unique watershed conditions.

4. Add 20 mL of standard solution to the cylinder in two 10-mL increments. Note the EC and temperature after each increment. The choice of standard depends on the expected range of the peak concentration measured in the stream. Concentrations at each step are the concentration above background.

- a. Range 1 = 0–40 mg/L using standard 1 at 2000 mg/L NaCl
 - i. Step 1 concentration = 20.2 mg/L
 - ii. Step 2 concentration = 40.0 mg/L
- b. Range 2 = 0–400 mg/L using standard 2 at 20 000 mg/L NaCl
 - i. Step 1 concentration = 20.2 mg/L
 - ii. Step 2 concentration = 40.0 mg/L

5. Calculate the CF for each step.
 $CF = \Delta(\text{Concentration}) / (EC - EC_{\text{base}})$

These readings should be tabulated and maintained over the long term. This will result in a record of temperature-specific CF measurements that can be used to derive a site-specific “CF curve.”

Equipment

The main pieces of equipment needed to measure streamflow by salt dilution are the electrical conductivity (EC) logger, scales for weighing salt, and a calibration kit for performing simple calibrations *in situ*. A notebook computer may be needed depending on the capability of the EC logger.

EC logger

We have used two different 4-electrode temperature-compensated EC probes with dedicated micro-loggers to collect our instream conductivity data. These instruments auto-range between 0 and 200 000 $\mu\text{S}/\text{cm}$, with an accuracy of $\pm 0.5\%$ of the measured value. The conductivity logger should be capable of measuring and recording stream

temperature as well as EC (primarily from 0 to 1000 $\mu\text{S}/\text{cm}$) at a log interval of 5 s or less. These instruments also have linear temperature compensation and data can be recorded in either compensated or raw (i.e., not corrected for temperature) form. However, we have found the temperature correction either unreliable or not useful since, as stated above, stream chemistry also affects the value of CF. Therefore we find that the raw EC data should be used and a specific value for CF should be derived through *in situ* calibration.

Scales

The accuracy of streamflow measurement is usually expressed as a percentage of the true streamflow. In this method, the accuracy to which the injection mass of salt can be measured is clearly an important factor. At low flow (0–2 m^3/s) using 1 kg of salt for 1% accuracy, the salt mass should be known to within 10 g. A good-quality scale for weighing food is probably sufficient for this level of accuracy. For low flows requiring less than 2 kg of salt, injection masses can be pre-weighed and kept in zip-lock freezer bags. The bags should be weighed before and after use to determine the actual mass of salt used in a measurement. For high flows ($> 20 \text{ m}^3/\text{s}$), salt can be added in multiples of 20 kg using the bags directly from the bulk distributor. The bags of salt should be weighed to within 200 g for 1% accuracy. Typically this level of accuracy can be achieved with a good-quality bathroom scale. However, for weighing salt masses suitable for probe calibration, a laboratory scale with a range of 0 to 2000 g capable of 2 digits of precision (i.e., $\pm 0.01 \text{ g}$) is required.

Application of Dry Injection at a Stream-gauging Site

A stream-gauging site that is not suited to current metering or other methods (weir, flume, etc.) might be

gauged by dry injection salt dilution. The following list outlines the criteria that should be considered before applying the method.

Preliminary criteria

Evaluate the site for suitability. The basic characteristics of a reach suitable for salt dilution are:

1. Turbulent at all flows.
2. Steep gradient: Some channels with gradients between 3 and 5% can be measured with salt dilution. Low gradient ($<3\%$) reaches tend to be suitable for current metering and high gradient ($>5\%$) for salt dilution.
3. Minimal pools and other backwater areas.
4. No tributary inflows in the gauging reach.
5. Riffle–pool, step–pool, cascade–pool morphology with cobble–boulder bed and flow constrictions.

The above criteria are easy to evaluate at a field site. However, the most critical considerations in applying the method are:

6. Ability to perform a clean injection at a point that favours mixing and rapid dissolution.
7. The salt must be fully mixed with the flow at the point where EC is measured in the channel.

To meet criterion #6 is simply a matter of technique. For “clean injection,” all the salt is injected into a point of turbulence with a single movement. The method may still work if the salt is injected in stages, but it will be difficult to determine from the dispersion graph if the measurement has been successful. The ideal injection point is a constriction in the channel where most of the flow converges and passes between boulders in the channel bed. For example, at Stephanie Creek a series of constrictions as the channel passes under the bridge (Figure 3) makes it an ideal injection site for two reasons:



R. Hudson

Figure 3. At Stephanie Creek the flow passes through a series of three constrictions that mix the salt very efficiently, resulting in a short mixing length.

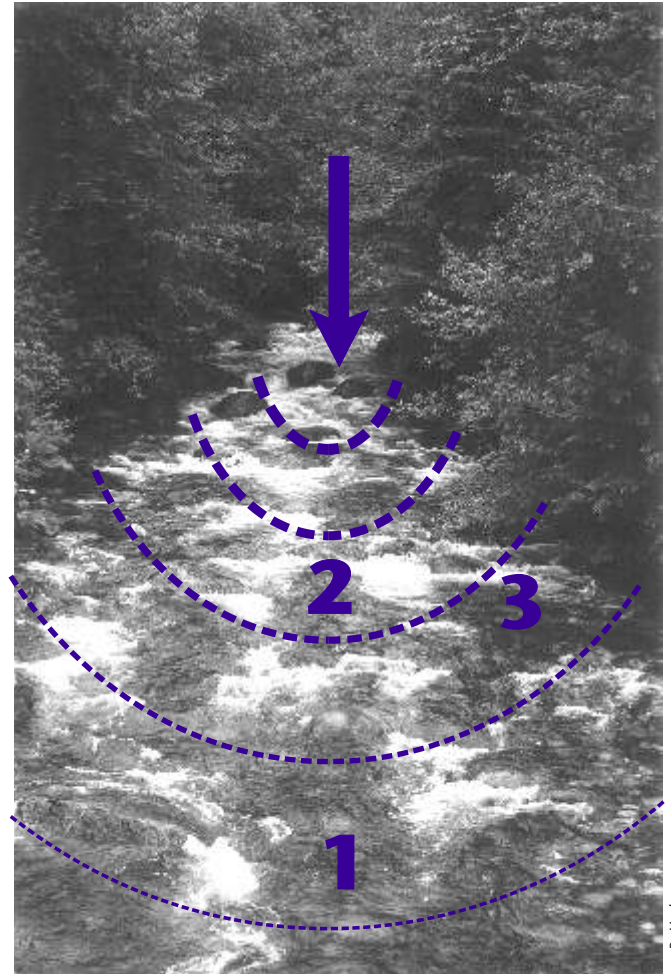
- the bridge can be used to dump the salt directly into the injection point; and
- the turbulence created below the constriction helps to dissolve the salt and mix the resulting solution into the water column.

Criterion #7 is more difficult to verify because it cannot be determined directly without deploying numerous probes in the channel. Choosing an appropriate mixing length is by far the most difficult aspect of the field procedure and requires an understanding of the dissolution and dispersion processes of salt in flowing water.

Behaviour of injected salt in a stream channel

The dissolution of salt in water takes time, with the rate of dissolution being proportional to water temperature (i.e., it dissolves faster in

warm than cold water) and inversely proportional to the existing concentration of salt. This dissolution behaviour can be easily observed: salt dropped into a glass of cold water will dissolve slowly because the water surrounding the grains has a high concentration, and tends not to mix with the water above it. However, once stirred the salt dissolves immediately. The rate of dissolution is more sensitive to concentration than to temperature. For dry injection at medium to high flows, the dissolution occurs at the lower concentrations. As noted previously, dissolution is greatly enhanced by a good injection point such that even in glacier-fed streams where water temperatures are in the 1–3°C range, it can be assumed to occur instantaneously. For low flows,



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Figure 4. Dispersion of salt at Russell Creek. Salt is injected at a point indicated by the arrow. The arcs represent the leading edge of the salt plume at successive time intervals as it disperses into the water column. The thickness of each line represents the salt concentration at each interval.

and particularly in wide channels with limited turbulence, the salt can be dissolved in a bucket of water before injection to aid mixing. This does not alter the method as long as the salt mass is known and fully dissolved in the water.

After injection, the salt mixes into the stream by longitudinal dispersion, a process in which dissolved salt in the plume moves along its concentration gradient until a uniform concentration exists. The dispersion process is superimposed on the flow (hence the term “longitudinal”), which means that the plume extends downstream faster than it does towards the banks (Figure 4). In this figure, the line that

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is farthest downstream represents a state where the salt plume approaches full mixing, making point 1 in Figure 4 the best location for the conductivity probe. At any time during the measurement the salt concentration is higher at point 2 and lower at point 3 than it is at point 1. Therefore, locating the probe at point 2 will yield a quantity A , which is too high, resulting in an under-estimate of the flow. Similarly, if the probe is at point 3, A will be too low and the resulting flow estimate too high.

Calibrating the Site: Mixing Length and Dosing Ratio

A conservative guideline is that the mixing length should be about 20 times the average channel width (Hudson and Fraser 2002). While Day (1977b) recommends mixing lengths of 25 times width (25XW), this is probably a conservative estimate since channel width is usually estimated visually. In most cases this can be assumed to be a "safe" mixing length. However, we have found that the optimum mixing length is often as low as 10 times the channel width, but users of the technique should conduct multiple trials to establish both the optimal injection points and mixing lengths for low and high flow at a particular site.

Determine the optimum mixing length

Since the rate of dispersion of the salt plume depends on several factors, each site will have a characteristic optimum mixing length. To determine this length, collect a sequence of measurements by varying the mixing length under stable flow conditions. The optimum mixing length is found where further increases in that length result in no relative change in the flow estimate.

Example of mixing length calibration

Culliton Creek is an ideal salt dilution site consisting of a long, straight reach with uniform gradient. The

morphology consists of a series of evenly spaced steps alternating with flow constrictions at each step. At

Table 1. Effective mixing length and resulting calculated flow at Culliton Creek

Discharge measurement	Probe and measurement	L (m)	Q (m ³ /s)
Q1	probe 2, M1	70	2.247
Q2	probe 1, M1	85	2.272
Q3	probe 2, M2	100	2.363
Q4	probe 1, M2	115	2.368

Culliton Creek two EC probes were used and two different injection points for 4 mixing lengths (Table 1; Figure 5). The channel is 10 m wide and the probes were deployed in mid-channel about 15 m apart. Two injection sites were chosen: the first site was about 70 m above the upper probe and the second about 30 m farther up the channel above the first injection site. This resulted in 4 measurements at increasing mixing lengths (Table 1).

This procedure shows that the minimum mixing length at Culliton Creek is 100 m, or 10 times the channel width. A standard methodology therefore in determining the optimum L might involve varying the mixing length from 10X to 20X the average width in 10- to 15-m increments. For high water, the relatively high flow velocity that typifies steep channels suggests that the mixing length should default to 20X the channel width.

Determine optimum salt dosing

To apply dry injection for flow measurement, the injection mass

must be known. This requires an accurate scale and a means of packaging salt for use in the field.

Zip-lock freezer bags can easily hold up to 2 kg of salt. Bulk salt (usually obtained in 20-kg bags from a bulk food distributor) can be pre-weighed into packages of various masses that can be made up in the field to the desired amount. These bags can be carried to remote sites in a backpack. Empty bags are weighed upon return to the lab or office to account for any residual salt not injected.

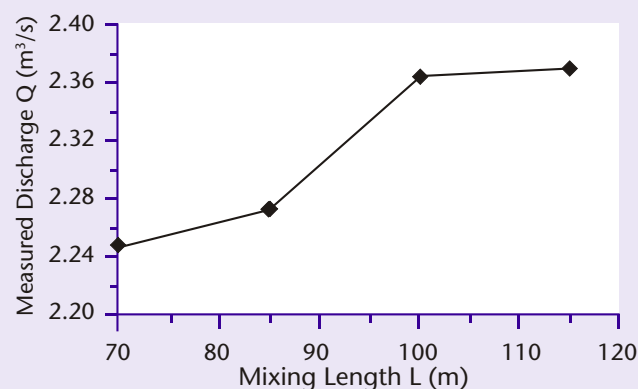


Figure 5. This chart shows how the increase in mixing length between each flow measurement results in an increase in measured flow to a maximum. The 100-m L is the optimum L .

The final step to determine salt dosing for a new site is to adjust the dosing mass needed to get a clean signal. In an earlier report (Hudson and Fraser 2002) we recommended dosing at the rate of 2 kg/m³/s of flow (dosing ratio = 2). Since then, we have found that in many cases the dosing ratio can be as low as 0.5. The aim of salt dosing is to add enough salt to get a clean signal without exceeding the most sensitive toxicity threshold (Moore 2004a, 2004b) of 400 mg/L NOEC (no observed effect concentration) for *Rana breviceps* (frog).

To get a strong signal, the difference between baseline and peak EC should be at least 100 times the resolution of the probe. Optimum dosing ratio is related to the optimal mixing length;

Table 2. A first approximation of dosing ratio in relation to the optimal mixing length (L)

Mixing length L (m)	Dosing ratio
10XW	0.5 – 1.0
15XW	1.5 – 2.5
20XW	2.0 – 3.0

the longer the mixing length, the more salt is needed. At a mixing length of 10XW, a dosing ratio of 1.0 works well. For longer mixing lengths, more salt is often needed for a clean EC signal (Table 2).

Limitations

Understanding the limits of applicability of dilution gauging methods in general will help to determine the appropriate technique for a given gauging site and whether to trust individual measurements. Operators should document their observations, thus contributing to a systematic assessment of the limitations of any technique and leading to informed decisions about their choice.

Here are some of our observations regarding the limits of applicability of the mass balance method:

1. Instream conditions: Turbulence is all-important. As a rule of thumb, conditions that violate the assumptions of current metering favour salt dilution and vice-versa. When applying the mass balance method with dry salt, try to observe the salt as it is injected. If it falls to the bottom of the channel and sits there in clumps, dry injection will not work. Either dissolve the salt or use constant rate injection or current metering. Some channel conditions render any method of measurement difficult or impossible. These include low flow conditions in very wide channels where the flow is dispersed by channel sediment.
2. There are situations where injected salt may be absorbed by (or may adsorb to) vegetation in

the channel or other substances (e.g., neoprene chest waders). Try to avoid applying salt dilution in channel reaches with a lot of aquatic vegetation. Always inject the salt in a downstream direction and, if possible, keep out of the channel during a measurement.

3. Violation of assumptions. In practice it is difficult not to violate some assumptions such as pools in the gauging reach. In riffle-pool channels, this is problematic since violation is a matter of degree — it can be minimized but seldom eliminated. If pools are large relative to channel area, then the salt will hang in the pool resulting in an extended tail. In some cases, pooling results in less than 5% error due to cutting off the tail of the distribution (Hudson and Fraser 2002) but the error will be systematic (i.e., it will tend to give an over-estimate).

4. Examine the conductivity-over-time graph. A clean dump with adequate dosing and full mixing has a characteristic shape. Many common problems can be detected this way:

- a. A smooth graph with a strong peak and short tail indicates a good chance of success.

b. Irregularities in the graph could indicate low dosing, improper mixing, or other problems that could render a measurement unreliable. For example:

- i. Double peak — is it lack of mixing or is it discontinuous injection?
- ii. Extended tail — does it indicate pooling of salt or a changing baseline?

This list of limitations is not comprehensive. The operator should recognize that documentation of observed limitations will lead to improved confidence in the application of dilution methods. There is no substitute for experience in applying salt dilution gauging. Over time an experienced operator will be able to judge the applicability of the method to a given site.

Comparison of Solution Injection and Mass Balance (Dry Injection) Methods

The mass balance method and slug injection of salt in solution method require similar calculations and both possess similar requirements for selecting a suitable measurement reach. Both methods also require full lateral mixing of the salt in the stream

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Table 3. Comparison of requirements of solution and dry injection methods

	Salt solution method	Dry injection (mass balance) method
Equipment requirements (not including EC logger)	Volumetric measurement equipment - 13 items (Moore 2005)	Weigh scale & packaging A bucket or similar for injecting Calibration kit (4 items)
Upper limits	Limited ability to measure high flow (upper limit 20–25 m ³ /s)	More-or-less full range of flows for small watersheds (i.e., up to 100 m ³ /s)
Calibration	Calibration of injection solution required for each measurement	Simple procedure to calibrate probe or to verify existing calibration. Required at each measurement for QA/QC
Low flow	Applicability depends on the channel morphology — for wide or braided channels at low flow, default to constant injection method (Moore 2004b)	Same as solution injection by dissolving the salt mass in streamwater before injection

and measurement of conductivity over time at a point downstream.

The mass balance method was developed for ease of application in the field. Solution injection with saturated (20%) solution requires the operator to manipulate a slug that is approximately five times more massive than a dry salt slug. In practice, a saturated solution is difficult to create in the field — a solution of 10–15% is more realistic. Using 0.5 kg of salt per cubic metre per second as a guideline to assess the upper limit of applicability and if the flow is 10 m³/s, dry salt will require the operator to inject at least 5 kg of salt while solution injection will require about 35 L, or approximately 35 kg, of solution to be injected. While this amount may be manageable, the ratio makes dry salt injection more appealing at higher flows. For example, we have measured flows greater than 40 m³/s with dry injection. To use solution to measure the same flow would require at least 300 L of solution. Thus the practical upper limit of applicability of the salt dilution method by solution is in the range of 5–10 m³/s. The use of dry salt instead of salt solution allows the upper limit to be extended to 100 m³/s or more.

Another advantage of the mass balance method is that it requires less equipment than the solution injection method (Table 3). The solution method requires the creation and calibration of injection solution for each measurement. This requires more time on site and more equipment to be carried to the field as well as more opportunity for error in calibration.

Salt solution has an advantage over dry injection in that it will mix more readily in lower, less turbulent flow, since the salt is already dissolved in water. However, the mass balance method can also be used with solution as a known mass of salt can be dissolved in a bucket of

streamwater before injection and the calculation carried out as though it were a dry mass.

Summary/Conclusions

Whether using solution or dry injection methods, the operator needs to have equipment for measurement and calibration. As long as the application criteria are met, it makes little difference which method is used. Both methods are capable of high precision. We have tested the mass balance method with dry injection under a wide range of conditions. Its speed and simplicity of application in the field make it an operational standard hydrometric method for small, steep streams.

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For more information, contact:

Rob Hudson, P.Geo., Ph.D., Hydrologist
613 Redwood Drive
Qualicum Beach, BC
V9K 1A2
Tel: (604) 341-0135
E-mail: Robert.Hudson@telus.net

John Fraser
Fraser Hydro-Technical
E-mail: Fras@telus.net

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Fertilization in Forested Watersheds

Robin Pike and Chris Perrin

In British Columbia, approximately 80% of municipal water supplies are derived from surface water sources. In these watersheds, it is not surprising that activities such as forest fertilization can raise public concerns and create perceptions of impaired water quality and aquatic ecosystem function. Between 10 and 30 years ago, several British Columbia watersheds were studied and monitored to address these concerns and quantify the effects of fertilization on water quality. While results have been widely available, they are mainly in the technical literature that is not commonly accessible to the public. Hence, many of the concerns raised in past decades about changes in water quality from forest and instream fertilization still remain today in many BC communities.

In this article, we revisit case studies of forest fertilization and provide a brief synthesis on the subject. The article focuses on forest fertilization and does not describe the effects of timber harvesting or application of herbicides, pesticides, or fire retardants on stream, lake, or soil water quality.

How Does Water Quality Vary Between Lakes and Rivers?

The natural or “background” quality of water is important to consider when predicting the potential effects of fertilization. Across British Columbia, water is generally of high quality, yet varies naturally in chemical composition. Water quality varies in time and space, and is a function of differences in bedrock geology, overlying vegetation, natural

disturbances, internal cycling processes, hydrology, and human disturbances.

Aquatic primary productivity in BC rivers and lakes is mainly limited by phosphorus (P), nitrogen (N), or the combination of both N and P. Across geographic zones, lakes and rivers are variable in nutrient content and are usually classified as eutrophic (nutrient rich), mesotrophic (moderate nutrient concentrations), or oligotrophic (nutrient limited). Seasonally, some rivers and lakes may experience natural spikes in nutrient load due to decomposing salmon carcasses, internal nutrient cycling (e.g., return of P from lake sediments to the overlying water), nitrogen-fixing species, and (or) seasonal flushing of accumulated nutrients from soils. In

most water bodies not affected by pollution, the main sources of N are from forest soils and the atmosphere, while P is mainly derived from mineral sources. In many regions, human activities appear to have dramatically added to these natural sources (Table 1). In Canada, household sewage is generally the largest point source of N and P discharge to the environment while agriculture is the largest non-point source of nutrient enrichment (Chambers et al. 2001).

Why Are Forests Fertilized?

Forest stands are most commonly fertilized to increase harvest volume and reduce rotation length. Fertilizers can also be applied to individual seedlings at the time of planting to minimize the time required to achieve “green-up” or “free-to-grow” or to release newly planted seedlings from competition for resources with other plants. In British Columbia, fertilization has been used in N-limited coastal Douglas-fir forests to increase productivity. It has also contributed to the renewal of cedar-hemlock and Sitka spruce growth that is in check (i.e., N and P limited). In the BC Interior, extensive

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Table 1. Comparison of P and N loading to Canadian surface and ground waters from various sources, 1996

Nutrient source	Nitrogen (10 ³ t/yr)	Phosphorus (10 ³ t/yr)
Municipality		
- waste water treatment plants	80.3	5.6
- sewers	11.8	2.3
Septic systems	15.4	1.9
Industry ¹	11.8	2.0
Agriculture ² (residual in the field after crop harvest)	293	55
Aquaculture	2.3	0.5
Atmospheric deposition to water	182 (NO ₃ ⁻ and NH ₄ ⁺ only)	n/a

Source: Chambers et al. (2001).

¹ Industry N loads are based on NO₃ + NH₃ and are thus DIN [dissolved inorganic nitrogen] not TN [total nitrogen that also includes organic forms of N]; industrial loads do not include New Brunswick, Prince Edward Island, and Nova Scotia. Quebec data are only for industries discharging to the St. Lawrence River.

² Agricultural residual is the difference between the amount of N or P available to the growing crop and the amount removed in the harvested crop; data are not available for the portion of this residual that moves to surface or ground waters.

research has found some soils to be sulphur (S) and N deficient (see Brockley 2000, 2004). For this reason, the majority of fertilizer used in this region includes a mixture of N and S to improve growth of lodgepole pine, spruce, and Douglas-fir.

From an economic perspective, fertilization of older immature stands is often a desirable investment, since treatment costs are compounded for only a few years before harvest. Fertilizers, however, are also applied to younger stands to ameliorate serious nutrient limitations and sometimes to accelerate stand development to mitigate expected mid-term timber supply shortfalls (R. Brockley, Research Silviculturist, Forest Productivity, BC Ministry of Forests, pers. comm., 2004). Data from the BC Ministry of Forests show that most operational forest fertilization on BC Crown land occurs in the Coast Region (Vancouver Forest Region) (Table 2).

Application of Forest Fertilizers

Fertilizer is most commonly applied at the stand level using a helicopter equipped with a centrifugal-deploying bucket and computer-assisted navigation. The effective swath of application is generally 60 m (30 m each side). Buffers (fertilizer-free zones) on streams are established to avoid deliberate deposition and (or) subsequent discharge of fertilizer into stream networks. While buffers are usually 10 m wide, a 40-m offset from the edge of the water body is usually necessary to attain the 10-m fertilizer-free zone and avoid the accidental deposition of fertilizer into streams (Perrin 1994).

In the past, fertilization has occurred in both the spring and in the fall, with spring often viewed as the optimal time. The nutrient demand of growing trees, combined with low average storm intensity and ideal temperature conditions for chemical

Table 2. Amount of fertilization¹ (ha) funded by all sources on BC Crown land, 1993–2003

Year	Area of Crown forest land fertilized (ha)						Total
	Cariboo	Kamloops	Nelson	Prince George	Prince Rupert	Vancouver	
1993/94	-	1,327	81	4	10	3,758	5,180
1994/95	11	493	966	528	138	4,732	6,868
1995/96	274	155	72	621	248	5,178	6,548
1996/97	922	189	2,520	824	1,302	14,263	20,020
1997/98	659	714	-	483	157	4,152	6,165
1998/99	1,671	187	21	468	93	11,669	14,109
1999/00	1	279	532	305	226	8,551	9,894
2000/01	898	139	54	237	47	11,960	13,335
2001/02	443	326	905	1,057	90	11,965	14,786
2002/03	44	263	19	219	10	3,924	4,479
TOTAL	4,923	4,072	5,170	4,746	2,321	80,152	101,384

Source: BC Ministry of Forests annual reports 1993–2003.

¹ Includes fertilization at time of planting but not fertilizer operations on private forestland.

conversions, makes spring a time when losses of fertilizer can be minimized from soils to surface drainages. However, volatilization (release of ammonia to the atmosphere) loss can be high at this time in some areas, making fall the preferred time in many areas.

Nitrogen is usually applied as prill urea (46-0-0¹) at rates of approximately 200 kg N/ha. Phosphorus is usually applied as mono- or di-ammonium phosphate (12-51-0) at a rate of 75–100 kg P/ha. Urea is generally used for Douglas-fir stands whereas urea with P is used in cedar–hemlock stands on northern Vancouver Island (G. Weetman, Professor Emeritus, UBC Faculty of Forestry, pers. comm., 2004). In the BC Interior, a blend of urea (46-0-0) and ammonium sulphate (21-0-0-24S) is used at application rates of about 200 kg N/ha + 50 kg S/ha. Further information on fertilization guidelines can be found in the *Forest Fertilization Handbook* (BC

Ministry of Forests and BC Ministry of Environment and Parks 1995)

<http://www.for.gov.bc.ca/tasb/legsregs/fpc/fpcguide/fert/ferttoc.htm>.

Why Are Rivers and Lakes Fertilized?

Parallel to forest management, some lakes, reservoirs, and streams in British Columbia are fertilized to increase primary productivity and subsequently the size and growth rates of fish. The added nutrients increase growth rates of algae, increasing the availability of food for zooplankton (in lakes) or benthic invertebrates (in streams and lakes). This, in turn, increases food availability for fish, yielding greater growth rates or survival. Fertilization of lakes with N and P has been used in the recovery of kokanee populations (Ashley and Slaney 1997; Ashley et al. 1999), following earlier successes from treatment of numerous sockeye nursery lakes in British Columbia and elsewhere (Hyatt et al. 2004).

¹ 0-0-0 = N-P-K = N = nitrogen, P = phosphorus, and K = potassium or potash expressed in percentages. For example, 25-4-11 = 25% N, 4% P, and 11% K.

Nutrient addition can also improve growth of trout in lakes (Johnston et al. 1999). Based on initial work by



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Perrin et al. (1987) and Johnston et al. (1990), stream fertilization is now a viable technique used in the recovery of coho and steelhead populations (Slaney and Ashley 1999). This strategy can kick-start the replenishment of nutrients to streams through the return of nutrients from salmon carcasses after spawning (Johnston et al. 2004). Wildlife, such as bears feeding on salmon carcasses, can also replenish nutrients in forests near salmon streams (Gende et al. 2002).

Aquatic Fertilizer Application

In freshwater ecosystems, P usually limits primary production, although “co-limitation” with N can occur when both are present in extremely low concentrations. Aquatic fertilizers can include 10-34-0 (ammonium polyphosphate) and, where applicable, 28-0-0 (urea – ammonium nitrate). Application rates vary with stream discharge/lake volume and background levels of N and P. For example, Big Silver Creek (Harrison Hot Springs) was fertilized in 1995 with 9.5 tonnes of ammonium polyphosphate, whereas application rates for the same year in the smaller Adam River on Vancouver Island were 2.3 tonnes (Slaney and Ashley 1999). Similarly, the application rate for lakes depends strongly on lake volume and management goal. Nutrients can be added to streams in liquid fertilizers,

as salmon carcasses, carcass analogue, solid fertilizer in prill form, solid briquettes, or using struvite

(magnesium-ammonium phosphate). Lake applications can involve aircraft although less expensive options such as using boats or barges are often preferred. See Ashley and Stockner (2003) for a complete review of aquatic fertilization protocols.



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What Are the Concerns with Forest Fertilization?

Numerous reviews have described potential effects of forest fertilization on water quality and biological production in aquatic ecosystems (e.g., Fredriksen et al. 1975; Moore 1975; Bisson et al. 1992; Binkley and Brown 1993; Perrin 1994; Binkley et al. 1999a, 1999b; Anderson 2001). These concerns are mostly related to unintentional increases in concentration of urea, nitrate, ammonia, ammonium, phosphorus, sulphur, and heavy metals in water bodies.

It is important to note that the term “ammonia” is often used interchangeably with the term “ammonium.” In this article, “ammonia” refers to the un-ionized and potentially toxic form of inorganic N (NH_3), while “ammonium” refers to the ionized form (NH_4) that is non-toxic in aquatic systems (Canadian Council of Ministers of the Environment [CCME] 2004). The term “total ammonia” refers to the combination of NH_3 and NH_4 . Both NH_3 and NH_4 can exist in water but NH_3 is only stable at pH greater than 9. Concentration of NH_3 decreases tenfold with each decrease in pH unit. Thus, for forest soils of 4–6.5 pH, only trace amounts of toxic ammonia are typically present (Binkley et al. 1999a). Differences in concentrations between the two forms are related to variations in pH and water temperature. The remainder of this article discusses, through case studies, observed increases in fertilizer-derived nutrients and compares these findings to current drinking water guidelines and criteria for aquatic life.

Nitrate and Total Ammonia

Increasing nitrate and total ammonia concentrations in water bodies are frequently one of the greatest public concerns in using fertilizers and are often perceived to cause increased primary productivity and (or) drinking water problems. After application, urea fertilizer in the presence of soil moisture is hydrolyzed to yield ammonia and then ammonium, which is effectively retained at cation exchange sites in the soil. That ammonium can be actively taken up and used as an N source by trees (Perrin 1994). The ammonium can then be nitrified in the presence of autotrophic bacteria common in BC soils to form nitrate (NO_3^-). This nitrate can also be taken up and used as an N source by trees, but it is relatively mobile in forest soils and can be lost to streams more so than ammonium. Soil temperature largely determines the rate of these

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transformations from urea to ammonium and nitrate (Perrin 1994).

Nitrate can be a concern where concentrations greater than 10 g/L as nitrate-N² adversely affect human health and can cause methemoglobinemia or “blue baby” syndrome in babies (Binkley et al. 1999a). In North America, background concentrations of nitrate-N in streams are low and average 0.31 mg/L with a median value of 0.15 mg/L (Binkley 2001). Concentrations are typically lower than this in coastal streams of British Columbia, sometimes reaching as low as 0.010 mg/L (Perrin et al. 1987). Nitrite (NO₂-) is a form of nitrogen that tends to be oxidized rapidly to nitrate in the environment, and thus thresholds of approximately 1 mg/L as nitrite-N in water bodies may never be reached (Binkley et al. 1999a). The guideline concentration for nitrate listed in the Canadian Environmental Quality Guidelines (CCME 2004) for community water supplies is 45 mg NO₃/L. The guideline concentration for protection of freshwater aquatic life is 13 mg NO₃/L for nitrate and 0.06 mg NO₂/L for nitrite.

NH₃ (ammonia) concentrations as low as 0.03 mg N/L can be potentially toxic to aquatic organisms in the short term and concentrations greater than 0.002 mg N/L (as ammonia) may be toxic over the long term (Binkley et al. 1999a, 1999b). While the CCME guideline for total ammonia varies with temperature and pH, the listed guideline for un-ionized ammonia is 19 g/L. Refer to the CCME guidelines (CCME 2004) for further details.

In the case studies reviewed, most fertilizer applications resulted in short-term increases in N (NO₃-N and total ammonia) concentrations to streams. When leave strips or buffers were applied to water bodies, relatively low peak concentrations of nitrate-N and ammonium-N were

found. Concentrations of total ammonia-N generally remained above background concentrations for several weeks to months after treatment whereas nitrate-N concentrations remained above background for several months to a year after treatment (Perrin 1994). Relatively high nitrate-N concentrations have been observed when fertilizer was directly applied to streams, under repeated fertilization, when using ammonium nitrate versus urea-based fertilizers and when fertilizing N-saturated hardwood forests (Binkley et al. 1999b). Overall, N concentrations following fertilization generally remained below the CCME guidelines (Table 3).

One case study in Table 3 is a notable exception. In the application of 224 kg N/ha of urea fertilizer to 80% of the Lens Creek watershed with no attempt to prevent direct deposition to water bodies or streams, Hetherington (1985) reported peak nitrate-N concentrations that approached the drinking water guideline. Hetherington (1985) reported increased concentrations of urea-N and ammonia-N that persisted for 7–13 days after fertilization, while nitrate-N levels remained above background for the duration of the 14-month study. High nitrate levels were explained as a result of nitrification during the dry 7-week post-application period, presence of red alder (from which nitrogen fixation can occur), steep slopes, and above-average precipitation (Hetherington 1985).

Three of the studies reported peak total ammonia-N levels beyond 1.98 mg/L possibly approaching toxic thresholds for aquatic life (although data for pH and temperature were unavailable to assess). Consistent among these studies was a lack of buffers around waterways. Although fish mortality was not specifically quantified as part of these studies,

none of them reported fish mortalities. Results of these case studies strongly suggest that avoiding direct deposition of fertilizer into water bodies is an important management practice to reduce the peak total ammonia concentrations in stream water. Indeed in British Columbia, the use of 10-m leave strips is a standard practice in all fertilization treatments to avoid aquatic toxicity (BC Ministry of Forests and BC Ministry of Environment and Parks 1995).

Urea

Excess urea reaching water bodies is less of a concern than transport of total ammonia. Urea is highly mobile in forest soils. Excessive leaching of urea can occur during heavy rainfall if it is not broken down into chemical forms that can be used by vegetation (Perrin 1994). There is no published toxicity guideline for urea-N. However, concentrations of several thousand milligrams per litre are required for any toxic effects, which would be much higher than what might reasonably occur in any stream environment (Binkley et al. 1999a). Surveying the literature, peak urea-N concentrations generally range from 0.1 to 50 mg/L within the first week after fertilization (Perrin 1994). Peak concentrations from various studies shown in Table 4 fall within this range. Case studies of treatments without buffers resulted in the highest peak concentrations, again showing the importance of avoiding direct deposition of fertilizer into water.

Phosphorus

Although the addition of phosphorus to lakes and streams is an effective fish management technique (described above), it is typically done with very tight control of the specific phosphorus load. Because P is often the nutrient that primarily limits biological production, overloading a system can lead to excess production

²Concentrations of nitrate and nitrite in water are often expressed in units of nitrate-nitrogen and nitrite-nitrogen, where 1 mg nitrate-nitrogen/L = 4.43 mg nitrate/L, and 1 mg nitrite-nitrogen/L = 3.29 mg nitrite/L.

Table 3. Peak nitrate-N and total ammonia-N concentration from BC case studies

Location	Treatment level (urea)	Buffer width (fertilizer-free zone)	Nitrate-N control ³ (mg/L)	Peak nitrate-N concentration (mg/L)	Duration ⁴ of elevated nitrate-N levels (days)	Total ammonia-N (NH ₄ + NH ₃ -N) control ³ (mg/L)	Peak total ammonia-N (NH ₄ + NH ₃ -N) (mg/L)	Duration ⁴ of elevated ammonia levels (days)	References
Sayward Forest ¹	200 kg N/ha	None	0.001	0.34	111	<0.004	3.36	111	Perrin et al. 1984
Sayward Forest ²	200 kg N/ha	50 m	0.001	0.19	100	<0.004	0.243	142	Perrin et al. 1984
Lens Creek	224 kg N/ha	None	0.028–0.151	9.3	435	0.08	1.9	13	Hetherington 1985
Gold Creek	176 kg N/ha with 52 kg S/ha	20 m	-	0.04	-	0.005–0.010	0.013	-	MacDonald 1987
Keogh River Tributary	200 kg N/ha	None	0.01–0.08	0.120	14	0.01–0.015	4.6	113	Perrin 1987a
Gold River	200 kg N/ha	30 m	0.02–0.07	0.28	73	0.011–0.017	0.03	3	Perrin 1987b
Slesse Creek	200 kg N/ha	10 m	0.21–0.33	0.21–0.33	n/a	0.020	0.024	2	Perrin and Bernard 1988
Glerup Creek	200 kg N/ha	10 m	<0.01	0.047	>39	<0.01	0.57	31	Perrin 1989
Glerup Creek	200 kg N/ha	None	<0.01	0.301	96	<0.01	3.69	31	Perrin 1989
Qualicum River	200 kg N/ha	30 m	0.035	0.088	136	<0.005	<0.005	n/a	Perrin 1990a
Rosewall Creek	200 kg N/ha	30 m	0.005–0.035	0.057	136	<0.005	<0.005	n/a	Perrin 1990a
Mashiter Creek	200 kg N/ha	50 m	0.005–0.01	0.037	47	<0.005	<0.005	n/a	Perrin 1990b
Tarundl Creek	200 kg N/ha	30 m	0.03–0.130	0.03–0.130	n/a	<0.005	0.160	29	Perrin 1991
Honna River	200 kg N/ha	30 m	0.022–0.097	0.022–0.097	n/a	0.01–0.02	0.042	9	Perrin 1991
Honna River Trib.	200 kg N/ha	None	<0.005	0.130	>97	<0.005	0.471	61	Perrin 1991
Nusatsum River	200 kg N/ha	None	0.005–0.035	0.088	>108	<0.005	<0.005	n/a	Perrin 1992
Sallompt River	200 kg N/ha	30 m	0.12–0.17	0.21	>141	<0.005	<0.005	n/a	Perrin 1992

Source: Perrin (1994).

¹ Data are the maximum concentration from 6 replicate streams, none affected by fertilizer spill.

² Data are the maximum concentrations from 4 replicate streams.

³ Data from sample sites not affected by fertilization and indicate concentrations at the time that peak concentrations were measured at the treatment site.

⁴ Number of days from time of fertilization to return to control values.

**Table 4. Peak urea-N concentration from BC case studies**

Location	Treatment level (Urea)	Buffer width (fertilizer-free zone)	Urea-N control ¹ (mg/L)	Peak urea-N concentration (mg/L)	Duration ⁴ of elevated urea levels (days)	References
Sayward Forest ¹	200 kg N/ha	None	<0.005	49.3	141	Perrin et al. 1984
Sayward Forest ²	200 kg N/ha	50 m	<0.005	0.658	136	Perrin et al. 1984
Lens Creek	224 kg N/ha	None	0.03	14.0	7	Hetherington 1985
Nusatsum River	200 kg N/ha	None	0.005–0.018	0.005–0.018	n/a	Perrin 1992
Sallompt River	200 kg N/ha	30 m	0.005–0.018	0.005–0.018	n/a	Perrin 1992
Tarundl Creek	200 kg N/ha	30 m	<0.01	0.614	66	Perrin 1991
Honna River	200 kg N/ha	30 m	0.005–0.020	0.296	10	Perrin 1991
Honna River Trib	200 kg N/ha	none	0.005–0.055	2.43	91	Perrin 1991

¹ Data are the maximum concentration from 6 replicate streams, none affected by fertilizer spill.

² Data are the maximum concentrations from 4 replicate streams.

³ Data from sample sites not affected by fertilization and indicate concentrations at the time that peak concentrations were measured at the treatment site

⁴ Number of days from time of fertilization to return to control values.

Table 5. Peak soluble reactive P concentrations from BC case studies (data from Perrin 1994)

Location	Treatment level	Buffer width (fertilizer-free zone)	Soluble reactive phosphorus (SRP) control ¹ (mg/L)	Peak soluble reactive phosphorus (SRP) concentration (mg/L)	Duration ² of elevated SRP (days)	Reference
Keogh River Trib (no buffer)	100 kg P/ha	None	<0.001	10.6	120	Perrin 1997
Glerup Creek (no buffer)	100 kg P/ha	None	<0.001	1.92	73	Perrin 1989
Glerup Creek	100 kg P/ha	10 m	<0.001	0.560	73	Perrin 1989

¹ Data from sample sites not affected by fertilization and indicate concentrations at the time that peak concentrations were measured at the treatment site.

² Number of days from time of fertilization to return to control values.

Continued from page 16

of unwanted algae and other water quality problems including reduced oxygen content, which kills fish, and may lead to drinking water impairment (taste/odor/treatment difficulties) (Wetzel 2001). The magnitude of the effect, however, depends on the characteristics of the receiving water body (e.g., discharge, background P concentration, water residence time).

Phosphorus itself is not toxic in drinking water and is not included in the CCME guidelines (2004) for community water supplies. Only a few case studies describe the extent of P loading caused by forest fertilization, mainly because most case studies have been conducted in association

with urea applications. Of the case studies available, peak soluble reactive phosphorus (SRP) concentrations range from 0.5 to 10 mg/L (Table 5), which is considered very high in aquatic ecosystems and well within the range that can increase growth of algae. These high concentrations, however, were not sustained and typically dropped close to, but still above background, within a few days after treatment thus limiting any time course change in biological production. Perrin (1994) found that low-level P concentrations dropped to background within 120 days, which means that some increased productivity may occur in streams receiving the P load over about four months. This low-level enrichment

may actually be a benefit for production of fish food in streams, particularly when conducted in the spring when water temperatures are rising and feeding by fish is active (Perrin 1994).

Sulphur

In areas where a blend of urea and ammonium sulphate fertilizers are used, concerns over increased S concentration in water may be raised. There is no maximum acceptable concentration for sulphate, but the CCME (2004) lists aesthetic objectives for drinking water of 500 mg/L and 1000 mg/L for livestock. The CCME guidelines also note the possibility of a laxative effect when sulphate concentrations exceed 500 mg/L. A

guideline of 500 mg/kg of S (elemental) is listed for the protection of soil under agricultural land uses. In preparation of this article, no case studies were located that studied post-fertilization increases in S levels resulting from forest fertilization.

Heavy Metal Concentrations

Because fertilizers can contain metals, the addition of fertilizers to water has been raised as a water quality concern. Urea fertilizers typically contain very low levels of heavy metal contaminants, whereas fertilizers containing P can have variable metal content (R. Brockley, pers. comm., 2004), largely related to the source that is mined to produce the fertilizer. The use of fertilizer databases (see text box) and chemical assays are two prudent ways to address concerns over potential heavy metal additions to the environment before application. Chemical assays can also verify that storage and distribution of fertilizer have not affected heavy metal concentrations post-production. In preparation of this article, no case studies were located that studied post-fertilization increases in heavy metal concentrations resulting from fertilization.

A useful source of information on fertilizer contaminants is available from the Washington State Department of Agriculture (WSDA) (<http://agr.wa.gov/PestFert/Fertilizers/ProductDatabase.htm>). This Web site provides information on concentrations of nine heavy metals (i.e., arsenic, cadmium, cobalt, mercury, molybdenum, lead, nickel, selenium, and zinc) in fertilizer. The Web site allows users to search by product name.

Is Fertilization in Forested Watersheds a Cause for Concern?

After a forest has been treated with fertilizer, changes in concentration of fertilizer elements can be expected in surface drainage. Case studies that form a technical basis for the present *Forest Fertilization Guidelines in British Columbia*, and that have been cited previously, suggest that unintentional nutrient transport to water bodies following forest fertilization is generally not enough to impair aquatic ecosystem health or affect the

Case studies suggest that unintentional nutrient transport to water bodies following forest fertilization is generally not enough to impair aquatic ecosystem health or affect the quality of drinking water.

quality of drinking water. Most of these studies reported increased post-fertilization levels of ammonia-N, urea-N, and nitrate-N for several days to months, but that these increases generally remained well below water quality guidelines.

Across the studies, the biggest increases were at sites where fertilizer-free zones were not used. These application issues have been addressed in recent years through mandatory buffers and restrictions on the percentage of a watershed that can be fertilized. Results of the case studies noted above strongly support the continuation of these key management practices.

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For further information, contact:

Robin Pike

Watershed Management Extension Specialist, FORREX
Victoria, BC

Tel: (250) 387-5887

E-mail: Robin.Pike@forrex.org
or

Chris Perrin

Limnotek
Research and Development Inc.
Vancouver, BC

Tel: (604) 222-3546

E-mail: cperrin@limnotek.com
www.limnotek.com

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Turbidity and Suspended Sediment as Measures of Water Quality

Paul Marquis

Turbidity is one of 82 chemical and physical parameters listed by Health Canada in their *Summary of Guidelines for Drinking Water Quality* (Health Canada 2004). Elevated turbidity levels are commonly associated with poor resource extraction practices as sediment from roads or disturbed riparian areas collects in runoff. As a measure of water quality, turbidity is frequently (and sometimes inappropriately) used as an analogue for estimating suspended sediment concentrations.

This article examines the connection between turbidity and sediment, and discusses some problems associated with the quantification of these variables. This article provides a general review of the subject. Data included in this article are cited for illustration only, and therefore detailed methodologies of data collection and analysis are not included.

Defining Turbidity

Turbidity is a measure of the cloudiness of a liquid and is usually quantified in nephelometric turbidity units (NTUs). Either organic matter, such as algae, or inorganic particles, like silt, can cause turbidity. Generally, water colour is not a good indicator of turbidity as dissolved compounds such as tannins can cause the water to appear dark without influencing its cloudiness. Turbidity is usually measured by passing a beam of light through a water sample and quantifying the scattering of the photons. Using these methods, turbidity can be measured very

accurately in the laboratory (e.g., to within 0.1 NTU) or with less precision under field conditions (Figure 1).

High turbidity levels can degrade the quality of drinking water by reducing the effectiveness of water-disinfection treatments, as the particles that cause turbidity are also vectors for pathogens. These particles can also

greater than 5 NTUs appears cloudy to the human eye, this value is commonly used as an aesthetic objective for water at the point of consumption (i.e., at the tap). British Columbia provincial guidelines consider 5 NTUs to be an acceptable level if it can be demonstrated that disinfection treatments are effective (BC Ministry of Environment, Lands and Parks 1997).

High turbidity values can also be associated with conditions that are harmful to fish. For example, particles in the water may irritate fish gills. In response to these stimuli, the gills produce a mucus-like substance that reduces the capacity of the membrane to exchange gases and causes stress to the fish. Furthermore, when these particles are deposited on the

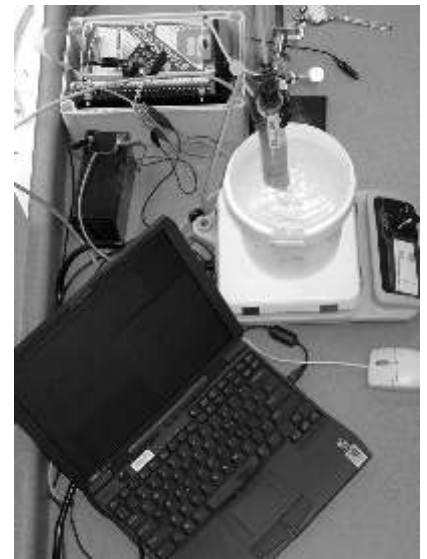


Figure 1. Left: A turbidity sensor installed in a natural stream channel. The OBS-3 probe was originally installed to specifications (D&A Instrument Company 1993), but a peak flow event changed the channel (as shown) and subsequently forced the relocation of this station. Right: An Analite 9500 turbidity probe set up in a laboratory. This brand of probe has a smaller "viewing window," which enables it to measure turbidity in a more confined environment.

shield pathogens from chemical treatments such as chlorination and from physical treatments such as ultraviolet irradiation. Health Canada has established guidelines limiting the amount of turbidity in potable water. These guidelines set 1 NTU as the maximum acceptable concentration for water entering a distribution system. Since water containing

streambed, they may cover spawning beds or sources of fish food such as benthic invertebrates. Establishing guidelines for aquatic life is more difficult as their susceptibility varies according to species, life-cycle stage, and the natural background turbidity levels of the water body. The BC Ministry of Environment, Lands and Parks (1997) has established turbidity

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guidelines for aquatic life. For example, the guidelines stipulate that under conditions of clear flows (i.e., where the suspended sediment concentration is less than 25 mg/L) induced turbidity should not be greater than 8 NTUs above the background level as measured in hourly samples collected over 24 hours. Other similar objectives have been established for longer periods and for watercourses with higher natural background turbidity levels.

Turbidity versus Sediment

Turbidity and sediment are not the same. Sediment refers to particulate matter moved by water and is typically measured as a ratio of these two components (e.g., milligrams of particulate matter to litres of water). Sediment is commonly divided into two groups: suspended sediment and bed load, which is also referred to as benthic sediment.

The size of the particulate matter in suspended sediment varies in proportion to the energy of the stream. Particles greater than 50 μm (i.e., sand) will fall out of the water column in seconds once the water is calmed. Silt-sized particles (50–2 μm) can remain in suspension for minutes in still water, while clay-sized particles (< 2 μm) can remain in suspension indefinitely.

Bed load usually refers to coarse sand and gravel-sized particles. This material is moved downstream by rolling or sliding along the bottom. It can also be transported through a form of movement called saltation, where particles skip or bounce by momentarily being pushed into the lower portion of the water column and then fall out again within a few seconds. Transportation of coarse sand and gravel requires higher stream energies and, as such, are usually associated with peak-flow events or high-gradient streams.

Quantifying the amount of sediment in streams involves both field

collection and laboratory procedures. Grab samples of suspended sediment can be collected either manually or with an automated system. The samples are then taken to the laboratory where the particles are removed by filtering and, after drying, the mass of the captured particles is determined on an accurate scale. Due to the variability of suspended sediment in natural watercourses, however, the grab sample approach may not provide a representative sample of the average or peak sediment concentration.

Developing a Relationship between Sediment and Turbidity

To obtain a more complete understanding of the sediment regime of natural watercourses, researchers have tried to establish a relationship between suspended sediment and turbidity (e.g., Jordan 1996; Hudson 2001a, 2001b). The relationship between these two variables is determined by measuring the turbidity of the grab samples at various suspended sediment concentrations and then applying this relationship to a data set of continuous turbidity measurements. For example, at 10:56 the data logger records a turbidity measurement of 100 NTUs and triggers the auto sampler to collect a grab sample. This water sample is then analyzed in the laboratory to determine the amount of suspended sediment it contains (in milligrams per litre), and this value is plotted against the recorded turbidity reading.

Once this procedure has been repeated several times over the entire

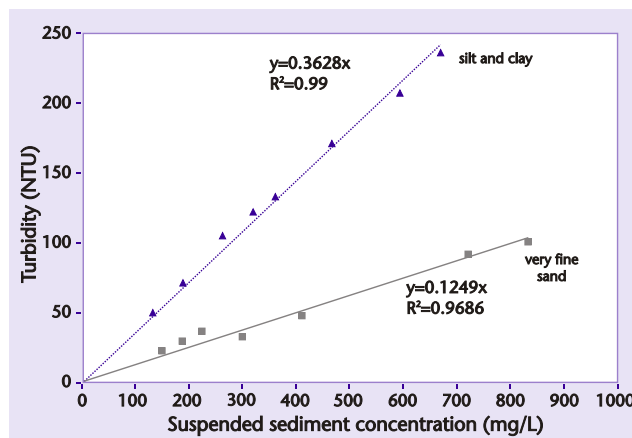


Figure 2. The dashed line represents the regression equation for a stream bank sediment sample that was sieved to eliminate particles greater than 53 μm . The solid line was derived for the same sample but for particles between 125 and 53 μm .

range of turbidity values recorded by the data logger, the suspended sediment/turbidity relationship can be expressed using a linear function (Figures 2 and 3) and applied to all of the data collected at the specific site. Maintaining the integrity of the suspended sediment/turbidity relationship, however, is difficult as the turbidity of a sample is not only influenced by its suspended sediment concentration but also by the size of the particles (Figure 2).

The suspended sediment/turbidity relationship is not explicitly transferable between watersheds.

The shape and composition of the particles also affect the turbidity of a sample. Figure 3 shows the regression lines of two different stream-bank samples (i.e., fluvial deposits) of very fine sand (i.e., particle sizes between 125 and 53 μm). The line with the greater slope was obtained from samples collected from Chinukundl Creek on the east side of Graham Island while the other sample was collected approximately 30 km to the west in Gregory Creek. The divergent slopes would tend to indicate that different geological origins and (or) processes are

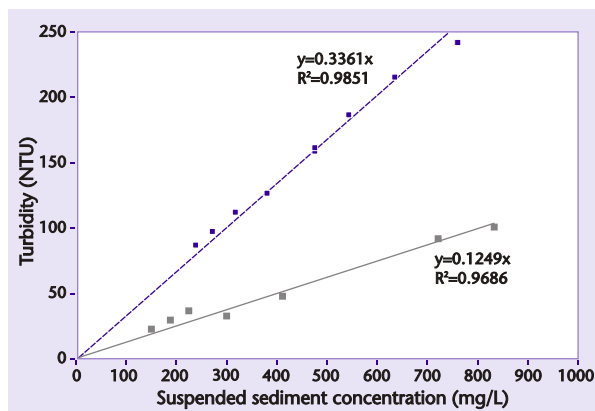


Figure 3. These two suspended sediment/turbidity relationships were developed using the same particle size category (i.e., very fine sand) but were collected from different watersheds. The disparity in turbidity for a given suspended sediment concentration may be attributable to differences in the shape of the particles or the composition of the parent material.

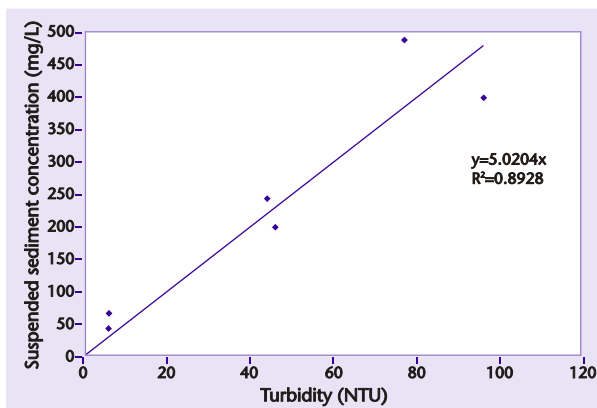


Figure 4. This suspended sediment/turbidity relationship is being developed for a study currently underway in the Queen Charlotte Islands. Note that the R^2 of 0.8928 is less than the value obtained for the similar relationships depicted in Figures 2 and 3, which were developed under idealized laboratory conditions. Before this function can be applied to the turbidity data, many more suspended sediment samples will have to be collected and analyzed.

responsible for sediment generation within the two watersheds. Unfortunately, it also means that the suspended sediment/turbidity relationship is not explicitly transferable between watersheds.

Furthermore, a specific suspended sediment/turbidity relationship may not even be valid over extended temporal intervals as the origins and proportions of sediment can change quickly during runoff. This problem can be further amplified by the hysteresis effect, in which finer

particles are more prominent on the rising limb of the hydrograph than they are on the recessional limb (Macdonald et al. 2003).

In reality, the close relationship between suspended sediment and turbidity as developed in the laboratory and depicted in Figures 2 and 3 is never realized with data collected in the field (Figure 4). Sources of error, such as algal growth on the optical components of the turbidity probe, air bubbles in the water column, changes in the composition of the suspended sediment, and variations in ambient light due to changing water levels, tend to corrupt the data. The use of turbidity as an indicator of suspended sediment can produce quality information, but often requires a large investment in instrumentation and much diligence from the investigator.

Summary

Although turbidity is a measure of water quality in itself, the causes of turbidity are more difficult to quantify. Furthermore, turbidity is usually the result of natural erosion processes. Therefore, using this measure as an indicator of the impacts of resource extraction is problematic. Nevertheless, measuring turbidity in conjunction with suspended sediment is likely to give a more comprehensive understanding of the factors affecting water quality than could be derived

by the examination of either of these measures in isolation.

For further information, contact:

Paul Marquis
Research Hydrologist
BC Ministry of Forests and Range
Coast Region
Nanaimo, BC
Tel: (250) 751-7051
E-mail: paul.marquis@gov.bc.ca

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UPDATE

Recent Publications and News

1. Selecting and Testing an Instrument for Surveying Stream Shade

by P. Teti and R. Pike.

Download the JEM article (798Kb) at:
www.forrex.org/jem/2005/vol6/no2/vol6_no2_art1.pdf

2. Post-Wildfire Watershed Rehabilitation in BC: Workshop Summary

Did you miss this workshop held on June 8, 2005? Presentations, posters, and an overall summary of take-home messages from the workshop are now available on the FORREX Web site:

www.forrex.org/programs/wildfire/

3. Carnation Creek Fish-Forestry Interaction Project Poster and Brochure

Two new publications from the Carnation Creek Fish-Forestry Interaction Project have recently been released. These publications (brochure and poster) summarize some of the long-term research results of the experiment and also outline the project's objectives, study components, and future directions.

"The effects of forestry practices on watershed processes and fish populations have been studied for 35 years at Carnation Creek. This intensive, single-watershed case study has generated the longest series of continuous data on fish-forestry interactions anywhere. Carnation Creek's long-term research approach has shown that key forestry-related changes continue to occur more than two decades after the main period of harvesting concluded, in spite of tree regeneration and growth throughout the watershed that is contributing to hydrologic recovery."

Brochure:

www.for.gov.bc.ca/hfd/pubs/Docs/Bro/Bro80.htm

Poster:

www.for.gov.bc.ca/hfd/pubs/Docs/P/P076.htm

4. Small stream channels and their riparian zones in forested catchments of the Pacific Northwest

A set of review papers published as a special issue of the *Journal of the American Water Resources Association*, Vol. 41(4), August 2005.

Guest Associate Editor:

R.D. (Dan) Moore¹

¹ Department of Geography and Department of Forest Resources Management, 1984 West Mall, University of British Columbia, Vancouver, BC, V6T 1Z2.
rdmoore@geog.ubc.ca

An ongoing concern in forest management is the risk associated with negative effects on riparian and aquatic ecosystems. The most common approach to protecting the values associated with streams and their riparian zones is the retention of riparian buffer strips and (or) special management zones. However, there are significant discrepancies among jurisdictions in the Pacific Northwest (PNW) about requirements for riparian protection along small headwater streams. Some jurisdictions require buffers that extend up to, and sometimes upslope of, the perennial channel network and include intermittent streams. Other jurisdictions do not require reserves along non-fish-bearing streams (unless they are a designated community water source) or even the smaller fish-bearing streams (Young 2000; Blinn and Kilgore 2001; Lee et al. 2004). The diversity of riparian management standards throughout the PNW has provoked increasing debate and highlighted the uncertainties in the current state of knowledge regarding small stream riparian systems and requirements for protection.

This special issue of *JAWRA* presents nine papers that review the current state of knowledge about the physical, chemical, and biological characteristics of small streams and their riparian zones as they relate to forest management in the PNW. These papers grew out of a symposium and workshop, "Small Stream Channels and Their Riparian Zones: Their Form, Function and Importance in a Watershed Context," held at the University of British Columbia in February 2002. These events were sponsored by the University of British Columbia, BC Ministry of Forests, BC Ministry of Sustainable Resource Management, Forest Renewal BC, Fisheries and Oceans Canada, and USDA Forest Service.

List of Article Titles and Authors

- Physical Hydrology and the Effects of Forest Harvesting in the Pacific Northwest: A Review
R. Dan Moore and Steve M. Wondzell
- Forest Harvesting and Streamwater Inorganic Chemistry in Western North America: A Review
Michael C. Feller
- Riparian Microclimate and Stream Temperature Response to Forest Harvesting: A Review
R. Dan Moore, David L. Spittlehouse, and Anthony Story
- Geomorphology of Steepland Headwaters: The Transition from Hillslopes to Channels
Lee Benda, Marwan A. Hassan, Michael Church, and Christine L. May
- Sediment Transport and Channel Morphology of Small, Forested Streams
Marwan A. Hassan, Michael Church, Thomas E. Lisle, Francesco Brardinoni, Lee Benda, and Gordan E. Grant
- Suspended Sediment Dynamics in Small Forest Streams of the Pacific Northwest
Takashi Gomi, R. Dan Moore, and Marwan A. Hassan
- Spatial and Temporal Dynamics of Wood in Headwater Streams of the Pacific Northwest
Marwan A. Hassan, Dan L. Hogan, Stephen A. Bird, Christine L. May, Takashi Gomi, and David Campbell
- Organic Matter Dynamics in Small Streams of the Pacific Northwest
John S. Richardson, Robert E. Bilby, and Carin A. Bondar
- Riparian Communities Associated with Pacific Northwest Headwater Streams: Assemblages, Processes, and Uniqueness
John S. Richardson, Robert J. Naiman, Fredrick J. Swanson, and David E. Hibbs

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