Peer-reviewed Research Article

Road Surface Erosion, Part 1: Summary of Effects, Processes, and Assessment Procedures

Elizabeth J. Baird, William Floyd, Ilja van Meerveld, and Axel E. Anderson

Introduction

Although resource roads provide economic and social benefits, they also pose a hazard to hydrologic, geomorphic, and ecologic processes (Trombulak and Frissell 2000; Gučinski et al. 2001). The ubiquity of roads in natural resource management areas makes understanding their potential impacts a fundamental element in assessing cumulative effects. Even well-designed road systems can alter streamflow and sediment budgets (Gučinski et al. 2001). Resource roads can increase mass failures and act as chronic or episodic sources of sediment through surface erosion (e.g., Beschta 1978; Bilby 1985; Rice and Lewis 1991; Luce and Black 1999; Fransen et al. 2001; Wemple 2001; Doten et al. 2006). By altering the rate and location of erosion and sedimentation, roads can affect hydrology and geomorphology, as well as negatively impact water quality and aquatic habitat. Roads change natural drainage patterns and alter the amount and distribution of overland flow (Croke et al. 2005). Roads also have the potential to change stream channel networks by concentrating overland flow, which can instigate the development of small channels and gullies, increasing drainage density and stream flow flashiness (Harr et al. 1975; Montgomery 1994; LaMarche and Lettenmaier 2001). In addition, roads can directly alter stream channel geometry at engineered road-stream crossings (Richardson et al. 1975), alter animal behaviour (Rost and Bailey 1979), act as barriers to animal and fish migration (Belford and Gould 1989), and facilitate the introduction of chemical contaminants and exotic species to a watershed (Trombulak and Frissell 2000).

Road sediment has been considered one of the major impacts to water quality and aquatic habitats (Reid and Dunne 1984). Resource roads have the potential to produce a
large amount of sediment, which is frequently delivered to stream networks at crossings (e.g., Reid and Dunne 1984; Luce and Black 1999; Megahan 2001; Baird 2011). This has led to the development of office- and field-based assessment methods designed to be applied over extensive areas to evaluate the hazard associated with stream crossings (e.g., B.C. Ministry of Forests 2001; Beaudry 2006; McCleary et al. 2007; Carson et al. 2009). These assessment methods are used to direct mitigation efforts and track environmental performance related to resource management goals. Directly measuring road sediment contributions to streams can be expensive and usually only provides detailed information from a limited area; however, these measurements can be used to test field-based methods and make necessary improvements.

This article is the first of a two-part series. In Part 1, we review road surface erosion and sediment delivery processes, along with road assessment methods that are commonly used in Alberta and British Columbia. In Part 2, we compare results generated by the Water Quality Effectiveness Evaluation protocol (Carson et al. 2009), a field-based assessment procedure used extensively in British Columbia, to results from a detailed study of surface erosion on an active logging road in a community watershed on Haida Gwaii.

The Effects of Road-derived Sediment on Streams

The effects of road-derived sediment on streams largely depend on whether sediment is deposited on the streambed or carried in suspension (Bilby et al. 1989). The probability that sediment is deposited on a streambed is controlled by the local shear stress divergence, which depends on particle size, flow rate, and the hydraulic properties of the channel (Bilby 1985). Coarse, sand-sized sediment (i.e., 62.5 μm and larger) is generally stored in small tributaries. More than 50% of road sediment can be trapped in small tributaries in some watersheds during normal precipitation events, with presence of large woody debris resulting in higher storage (Bilby et al. 1989). Increased sediment deposition in streams can infiltrate into and cover existing riverbed habitat, reducing suitable areas for organism growth, and altering invertebrate populations (Beschta 1978; Ramos-Scharron and MacDonald 2007). For example, increased sedimentation in small mountain streams may clog gravels used as fish spawning areas and reduce survival rates following spawning (Phillips et al. 1975; Beschta 1978; Tappel and Bjornn 1983). Preferred salmon spawning gravels, however, tend to be located at the top of riffles in areas of the stream with relatively high flow velocities. Sediment derived from nearby roads is usually too fine to be deposited in these areas (Bilby 1985).

Smaller, silt-sized particles (i.e., 62.5 μm and smaller) remain in suspension and are more likely to reach higher-order streams. These particles are thought to pose the greatest threat to aquatic environments (Fahey and Coker 1992; Ramos-Scharron and MacDonald 2007), although their effects are generally more subtle than those observed for coarser sediment (Bilby 1985). Wash load and suspended sediment reduces the amount of light available for photosynthesis, which can trigger impact cascades through many trophic levels (Newcombe 2003;
Ramos-Scharron and MacDonald 2007), and can also harm the development of fish eggs and larvae, alter fish migration, limit visibility for hunting/feeding, and irritate fish gills (Bilby 1985; Newcombe 2003; Marquis 2005). The consequences of wash load and suspended sediment on fish are a function of the concentration and duration of increased sediment concentrations (Singleton 2001; Newcombe 2003).

Stream ecosystems may be seen as resilient to large infrequent rainfall events (the primary generators of significant sediment transport) as these events do not necessarily have long-term effects (Fransen et al. 2001). Although few studies have confirmed adverse effects of sediment from resource roads on nearby stream environments (e.g. Fransen et al. 2001), some studies suggest that both deposited and suspended road-derived sediment can have long-term negative impacts on aquatic ecology and water quality, especially in watersheds with chronic fine sediment delivery (Beschta 1978; Bilby 1985; Newcombe 2003; Marquis 2005; Ramos-Scharron and MacDonald 2007). Pike et al. (2010) provided an in-depth review of water quality effects of elevated sediment concentrations.

Fine sediment can act as a vector for pathogens in drinking water because it reduces the effectiveness of water treatment by shielding pathogens from both chemical (e.g., chlorination) and physical (e.g., ultraviolet irradiation) disinfection (Marquis 2005). Health Canada has set the maximum turbidity\(^1\) of water entering a distribution system to no more than 1 Nephelometric Turbidity Unit (NTU), with a treated water target of less than 0.1 NTU (Marquis 2005; Federal-Provincial-Territorial Committee on Drinking Water 2010). Water quality guidelines in British Columbia for induced suspended sediment and turbidity in rivers are based on increases above background levels (Pike et al. 2010). For watersheds with sensitive ecosystems or drinking water facilities, sediment concentrations in rivers should thus be carefully monitored and sediment production from resource roads should be managed where it has the potential to reach streams.

**Surface Erosion and Sedimentation Processes**

Erosion is the process of detaching, transporting, and depositing sediment (small fragments of organic or inorganic material). Overland flow is the primary mechanism responsible for surface erosion of resource road sediment. The power of flowing water, which is the product of the specific weight of water, volume of flow, and the energy slope, controls the transport and deposition of sediment (Hairsine and Rose 1992a, 1992b).

The potential sediment sources in a forest road prism (Figure 1) include the road surface, cutslopes, fillslopes, and (or) ditches (Spinelli and Marchi 1996). Arnáez et al. (2004) found the cutslope of a road prism had the largest erosion rates, with gradient being the most significant control. This was attributed to mass wasting and freeze-thaw processes along the cut banks continuously supplying loose material for transport. Plant cover and stone cover density of the cutslope area also controlled erosion rates. However, Reid and Dunne (1984) found the cutslope, fillslope, and ditches contributed only a small amount of sediment compared to the road surface.

Vegetation and forest floor are removed during road construction. This exposes and loosens the soil, decreasing its cohesion and making it more susceptible to erosion. Following construction, compacted road surfaces have lower infiltration capacities than those of the surrounding landscape (Spinelli and Marchi 1996), which causes infiltration-excess overland flow, even for low-intensity precipitation events (Croke et al. 2005). Resource roads can also intercept subsurface flow, redirecting it over land (Spinelli and Marchi 1996).

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\(^1\) Turbidity refers to the degree to which transparency of a liquid is lost owing to the presence of suspended particulates; turbidity is thus an indirect measure of the concentration of suspended solids.
If large particles (> 2 mm) are present on the cutslope, fillslope, or ditches, an armour layer may develop and reduce the amount of sediment available for erosion. The presence of vegetation on cut slopes and fillslopes and in ditches greatly reduces the amount of coarse sediment available for erosion; however, the effects of vegetation on availability of finer sediment are not well documented (Luce and Black 1999). Removal of vegetation and disturbance of soil and armour layers following road maintenance generally leads to a temporary increase in the amount of sediment available for erosion (Luce and Black 2001).

Controls on Erosion
Climate, Soil Texture, and Vegetation Cover
The amount of sediment generated from a section of road is ultimately determined by the capacity and competence of the water flow that occurs across it. The frequency and intensity of precipitation affects sediment generation from road surfaces. Surface erosion rates, therefore, vary with climate (Gucinski et al. 2001; Thurton et al. 2009). Roads in wetter climates tend to have greater connectivity with streams, as buffers become saturated, leading to a reduced ability for surface runoff to infiltrate (Eliot et al. 2009).

If exposed soil contains large, gravel-sized particles (i.e., 2 mm or larger), preferential erosion of fine particles creates an armour layer, which reduces the erosion potential. Re-establishing and maintaining vegetation on cut slopes, fillslopes, and ditches further reduces the amount of coarse sandy sediment (i.e., 0.5 mm or larger) available for erosion (Luce and Black 2001; Grace 2002; Arnáez et al. 2004; Elliot et al. 2009; Thurton et al. 2009). Once armour layers and vegetation cover is established, limiting disturbance to soils during maintenance and use is key to prevention of erosion.

Road Design, Construction, and Maintenance
Longer road segments usually yield more sediment as these segments have a greater sediment supply and concentrate more water. Steeper road segments can transport larger particles owing to the increased flow energy, which leads to greater sediment yields (Bilby et al. 1989). Road crossings act as the main points for sediment to enter streams (Taylor et al. 1999). The spacing of road crossings, as well as their location and design, can affect the amount of road runoff directly entering a stream. To minimize the amount of sediment and runoff directly entering a stream, ditch water should be diverted onto the forest floor before reaching a stream crossing, and the road surface should be elevated over crossings so surface runoff travels away from the stream (see Gillies 2007). When roads are used intermittently, partial deactivation, including the installation of water bars to divert water from the road surface, can reduce erosion and lengthen the life of the road.

Resource roads produce the largest amount of sediment during the first 2 years after construction (Akay et al. 2008). The decline in sediment generation following construction is attributed to the establishment of armour layers and the re-establishment of vegetative cover along the cut slope, fill slope, and ditch areas. Road maintenance, which is necessary to keep conditions suitable for travel and prevent failure of drainage systems resulting in severe erosion, may create road conditions similar to those following initial construction (Fahey and Coker 1992; Luce and Black 2001); cleaning or grading ditches and removal of cut slope vegetation substantially increases sediment production. However, some studies have found grading only the road surface does not produce a statistically significant increase in sediment yield (Luce and Black 1999).

Traffic
Traffic can affect the amount of sediment produced from a resource road in many ways. For instance, fine sediment on road surfaces may be derived from the breakdown of surface material as vehicles pass and the forcing upward of fine-grained sediment from the road bed as traffic pushes the surface material into the bed (dynamic pumping) (Reid and Dunne 1984). Fine sediment may also be introduced onto the road surface to help bind road material together or may be a component of the material used to build the road. Fine sediment reduces the infiltration capacity of the road surface, increasing overland flow and erosion.

Traffic also causes cross-slope flattening, which directs water down the road surface rather than taking a direct route to the ditches (Foltz 1996). Roads with high traffic intensity or those whose surfaces are not well maintained may progress from cross-slope flattening to rut development. Flow concentrated in ruts has higher shear stress, which increases its ability to erode and carry sediment, thus augmenting erosion (Foltz 1996). For some resource roads, passage of more than four trucks per day is considered heavy traffic and may result in cross-slope flattening or rutting (e.g., Reid et al. 1981).

Connectivity
Connectivity between a road and stream describes the probability that runoff from a road, and the sediment it carries, will reach the stream network (Croke et al. 2005). Sediment is carried by runoff in either dispersive
or advective (channelized) pathways (Takken et al. 2006). Advective pathways are generally associated with culvert pipes; dispersive pathways are associated with mitre drains and push outs. Advective overland flow has little opportunity to deposit finer-grained sediment, whereas dispersive pathways may provide conditions for deposition (Lane et al. 2006). Most fine, silt-sized sediment (i.e., 62.5 μm and smaller) is carried as wash load, and will not settle out of suspension until the water infiltrates the soil (Hairsine et al. 2002). Dispersive flow is more likely to infiltrate before reaching a stream, as advective flow can travel two to three times further from the road prism prior to infiltration (Croke et al. 2005). Roads closer to streams are more likely to have higher connectivity because there is less distance for water to infiltrate and sediment to be deposited (Bilby et al. 1989; La Marche and Lettenmaier 2001). These processes are documented, but little is known about changes in sediment fluxes as runoff moves across the landscape to streams (Croke et al. 2005).

Erosion Potential Assessment Procedures

Empirical models predicting soil loss (e.g., the Universal Soil Loss Equation; Wischmeier 1976), and process-based prediction models (e.g., the Water Erosion Prediction Project; Nearing et al. 1989) have been developed for use in agricultural and forested watersheds. However, physically based soil-loss models tend to be highly parameterized, making application in poorly instrumented areas difficult. Sediment budgets can also be determined through measurements of stream water quality or deposited sediments. Although these measurements provide detailed information, they can be capital and labour intensive (e.g., Andrews 2006).

To direct mitigation efforts, the performance of resource management goals should be measured and the effects of roads on water quality should be determined for large areas. This need has led to the development of several office- and field-based methods to estimate the potential impacts of resource roads on water quality and aquatic habitats (e.g., Flanagan and Nearing [editors] 1995; B.C. Ministry of Forests 2001; Beaudry 2004; Mc Cleary et al. 2007; Carson et al. 2009). Based on detailed studies and a physical understanding of erosion processes, these procedures can be applied over large areas.

Office-based Procedures

A major advantage of many office-based procedures is the ability to assess potential sediment sources over large spatial scales. Air photos can be used to identify road-related sediment sources, including mass failures, unvegetated or unstable fillslopes and cutslopes, road surfaces with steep grades, and road sections close to streams or with high connectivity (B.C. Ministry of Forests 2001). The use of GIS to calculate stream crossing densities and highly parameterized erosion models such as the Water Erosion Prediction Project, along with emerging technology such as LiDAR used to identify sediment sources, are all becoming more common as the availability and usability of these resources improves. Nevertheless, these resources generally provide limited information on the actual amount of sediment that specific areas contribute and the consequences for water quality. These resources also do not provide any guidance on improving the design, construction, and maintenance practices for roads.

Direct Measurements

Sediment concentrations in road surface runoff can be measured using grab samples from road and ditch water (e.g., Bilby 1985). Settling traps and settling basins that intercept road and ditch runoff may also be used; however, it is difficult to determine how much suspended sediment passes through (e.g., Luce and Black 1999). Rainfall simulations can determine the relationship between road surface sediment generation and rainfall parameters, providing detailed site-specific information (e.g., Hamed 2002). Direct measurements of sediment concentrations in road surface runoff can be informative, although it may be difficult to actually determine how much of this sediment reaches surface waters and the overall impact it has on water quality.

Stream sediment concentrations can be directly measured using grab samples or pump samplers, or by using turbidity measurements as a proxy (e.g., Andrews 2006). Nevertheless, monitoring sediment yield at a single location within a river does not effectively differentiate road versus other sediment contributions. Although monitoring above and below crossings can isolate sediment contributions from a section of road (Kahhlnen 2001), instrumentation to continuously monitor stream sediment fluxes can cost $10 000–15 000 per crossing, which limits widespread application. Manual grab samples, while more economic than continuous monitoring, are generally ineffective at describing the effects of roads on streams because of the dynamic nature of river conditions and the necessity of concurrently collecting samples at many locations. Measuring the amount of road sediment in streambed deposits has also been used to show the influence of road sediment on streams (e.g., Bilby 1985; Spillois 1999).

Isolating sediment sources at larger scales becomes more difficult and directly measuring sediment delivery from roads and sediment concentration

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in streams is therefore not practical for most watersheds. Direct measurements can, however, be used to test and validate office- and field-based assessment procedures. If climate, traffic use, streamflow, and turbidity data is available in a watershed, statistical models can be developed to identify the main factors that increase turbidity at downstream locations.

Field Assessments
To estimate the volume of sediment a road may contribute to a stream, field assessments often use indicators of erosion potential instead of direct measurements of erosion. These procedures require sampling of 90–100% of crossings within a study area (Beaudry 2004), and are therefore generally designed to be quick, systematic, and objective. Field assessment methods usually incorporate physical characteristics of the site, such as slope and soil texture, proximity to a water source, and amount of vehicle use.

The Erosion Potential Index (Anderson and Anderson 1986) is used to evaluate the amount of erosion expected from the road prism surrounding a stream crossing based on measurements of slope gradient, slope length, disturbed area, and vegetation cover. This index was developed in Alberta, and has been used to determine whether crossings require reclamation work and whether erosion control strategies are successful. Although this process is similar to more recent field assessments, it does not consider each component of the road separately and does not take connectivity into account.

The Foothills Stream Crossing Partnership has also developed a procedure to estimate sediment delivery to a stream (McCleary et al. 2007). This procedure was designed for streams within Alberta where permanent roads cross defined channels. The area and vegetation cover are measured for all sediment sources that contribute to a stream crossing, such as roads, cutslopes, and ditchlines. A simplified version of the Revised Universal Soil Loss Equation is then used to estimate a sediment yield (McCleary et al. 2007).

The Stream Crossing Quality Index (Beaudry 2006) assesses the erosion potential of ditches, fillslopes, and road surfaces running into a crossing by considering the size, soil texture, slope gradient, and percentage of non-erodible cover of the sediment source, as well as the level of road use and ditch shape. The delivery potential of each component is assessed by estimating connectivity based on distance, slope, and potential for sedimentation prior to entering the stream network. Stream crossings are assigned a hazard rating based on a score derived from the site characteristics. This index has been applied extensively across British Columbia, primarily in the Central Interior and northeast of the province, and in northern Alberta.

The Water Quality Effectiveness Evaluation (Carson et al. 2009) was designed to estimate the potential contribution of fine sediment from stream crossings and other sources, such as existing mass failures, to a stream network on an annual basis. This protocol does not aim to establish an absolute volume of sediment delivered to each crossing but instead determines a rating based on sediment delivery categories. The stream width at the crossing may be used to estimate the stream sediment dilution potential to assign a rating when downstream water quality impacts are considered important. Ratings range from very low to very high.

The primary benefit of using indicators is the guidance they provide in improving water quality by identifying systematic problems with road planning and maintenance. Indicators also help to locate problem areas within a watershed and thus enable the development of mitigation strategies for these areas. Figure 2 shows an example of a rating map developed using results of a Water Quality Effectiveness Evaluation in the Naka Creek watershed on Vancouver Island. The rating map pinpoints the locations of the main sediment sources (large concentration of big circles) and high-impact areas (large concentration of red circles). In this example, high-impact areas were not always associated with high sediment delivery areas, as the ability of a stream network to dilute sediment was considered as well.

Summary
Resource roads pose a hazard to streams, predominantly through the introduction of road sediment, which can alter hydrologic, geomorphic, and ecologic processes, and negatively affect water quality and aquatic habitat. Increased sedimentation in streams can clog gravel used as fish spawning areas, reduce survival rates following spawning, and alter invertebrate populations. Elevated fine sediment concentrations, especially in watersheds where such conditions are chronic, can also have long-term negative effects.

Erosion and sediment delivery from roads to stream networks depends on climate, soil texture, vegetative cover, road design and maintenance, gradient, and traffic, making it inherently difficult to measure and predict the volume of sediment generated. Office-based procedures can be used to identify areas with potential to deliver sediment to stream networks, but these
procedures generally require field-based methods to identify site-specific problems. Field assessments used to estimate potential impacts on streams are generally quick, systematic, and objective. These assessments are commonly used to measure the performance of resource management goals or to direct mitigation efforts, and can be a useful tool in determining cumulative impacts in watersheds with multiple land uses. Field-based methods will continue to evolve as they are tested and applied in new areas.

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Introduction

In Part 1 of this series, we reviewed road surface erosion and sediment delivery processes, along with road assessment methods that are commonly used in Alberta and British Columbia. Here, in Part 2, we compare results generated by the Water Quality Effectiveness Evaluation (Carson et al. 2009), a field-based road assessment procedure used extensively in British Columbia, to results from a detailed study of surface erosion on an active logging road in a community watershed on Haida Gwaii.

Background

The Village of Queen Charlotte has recently opened a drinking water intake and treatment plant on the Honna River. Several hydrology-related projects have been conducted on the Honna River (e.g., Bruce and Chatwin 1987, 1988; Dobson Engineering Ltd. 1996) and have identified the Queen Charlotte Mainline resource road (hereafter referred to as the “QC Main”) as a chronic sediment source. Because elevated sediment concentrations can affect drinking water treatment processes, this project was initiated in 2009 to provide answers to the following questions.

Editor’s Note: This article is the second in a two-part series on resource road surface erosion and methods of assessing water quality impacts (this issue).

• How much does the road contribute to the overall sediment budget in the river?
• What periods of the year are of greatest concern to water quality?
• Where are potential sources of sediment from the QC Main located?
• What mitigation strategies are necessary?

Focussing resource road improvement and restoration efforts on road sections producing the most sediment within a watershed is an effective method to decrease overall road impact (Wemple 2001). The Water Quality Effectiveness Evaluation (WQEE; Carson et al. 2009) provides a method to estimate the potential contribution of fine sediment from stream crossings to a stream network on an annual basis. As such, it can potentially identify the portions of the QC Main that deliver high amounts of sediment to the Honna River. Although the WQEE has been applied extensively across British Columbia, it has undergone minimal quantitative testing to validate the estimated sediment volumes delivered at individual crossings. The objective of this article is to compare estimates of sediment volume and ratings generated by the WQEE with volumes calculated through a targeted experiment using a rainfall simulator during active log hauling, and turbidity data collected above and below two stream crossings.

Methods

Study Site

The Honna River watershed is located on Graham Island, which is part of Haida Gwaii, British Columbia. It is 4 km northwest of the Village of Queen Charlotte and approximately 30 km northwest of the community of Sandspit (Figure 1). The watershed is about 52 km², has a siltstone and mudstone dominated lithology, and extends from mean sea level to approximately 1000 m in elevation, with a north–south valley along the Honna River. The Honna watershed is located within the hypermaritime subzone of the Coastal Western Hemlock biogeoclimatic zone, one of Canada’s wettest and most productive forest regions (Meidinger and Pojar 1991; Egan et al. 1999). Temperatures in this zone are moderated by the Pacific Ocean, resulting in cool summers and mild wet winters (Meidinger and Pojar 1991). Long-term climate data are not available for the Village of Queen Charlotte, but climate normals are expected to follow the same trends as Sandspit, although the Honna Watershed receives more precipitation owing to orographic effects. On average, Sandspit experiences 222 days of detectable
precipitation (> 0.2 mm) in a year and has a daily average temperature of 8.3°C. For this study, precipitation was measured in the lower portion of the Honna watershed near kilometre 10 of the QC Main, as well as near Stanley Lake in the upper western portion of the watershed. Total precipitation between October 1, 2009 and May 31, 2010 was 1980 mm near kilometre 10 and 2212 mm near Stanley Lake, compared to 999 mm for Sandspit.

The road network within the Honna watershed is extensive; however, the QC Main is the major source of chronic road-related sediment because of its high amount of vehicle traffic and proximity to the Honna River. It is the primary route for off-highway logging trucks to the logging sort located west of the Village of Queen Charlotte. It is also the main access route to the west coast of Graham Island for recreational vehicles. Steady vehicle use is evident throughout the year due, in part, to the mild winters. During periods of hauling in the fall of 2009, the road experienced between 10 and 20 passes of loaded logging trucks per day, as well as light vehicle traffic (i.e., cars and pickup trucks). The QC Main closely parallels the Honna River between kilometre 3 and kilometre 10 (Figure 2), resulting in high connectivity potential between the road and river. The WQEE assessments and targeted rainfall simulations therefore focussed on this section of the QC Main.

Field Methods
Three different methods were used to determine the potential for road-related sediment to affect water quality in the Honna River:
1. the Water Quality Effectiveness Evaluation,
2. rainfall simulations during active hauling, and
3. turbidity monitoring above and below two crossings on the QC Main.

Water Quality Effectiveness Evaluation
In October 2009, 52 crossings (44 culverts and 8 bridges) along the 7-km study section of the QC Main were assessed using the WQEE protocol (Baird 2010). The road components (cutslopes, fillslopes, ditches, and road surface sections) contributing water to each crossing were identified. For each component, the area was measured by pacing, slope was measured using a clinometer, and the amount and texture of exposed soil was estimated using the WQEE guidelines. The connectivity of each component was quantified using estimates of the distance to the stream and the contributing area. Based on this information, a potential volume of sediment delivered to the stream was estimated and a rating for each road crossing was assigned (Table 1). See Carson et al. (2009) for a detailed description of the WQEE protocol.

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Figure 1. Location of the Honna Watershed in relation to the Village of Queen Charlotte. Turbidity probes are represented by yellow circles, with Honna kilometre 7, Honna kilometre 6, and Drinking Water Intake representing probes in the river’s main stem. Rain gauges are represented by orange circles.
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Figure 2. Locations of streams, roads, and stream crossings along the QC Main in the Honna Watershed with the WQEE rating (size of circle) representing the predicted amount of sediment contributed to nearby streams. Turbidity was monitored above and below crossings 16 and 28. Rainfall simulations were completed near crossing 40.
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Rainfall simulation

To quantify the erosion potential from the road surface, 24 large-scale (30 m by 4.5 m) rainfall simulation experiments were conducted near kilometre 8 of the QC Main in fall 2009 (Figure 3). The 5–50 mm/hr rainfall simulation experiments were 1–4 hours in duration and were used to determine the influence of rainfall intensity, rainfall amount, traffic intensity, and truck speed on total sediment production from the road. For a detailed description of the rainfall simulation experiments see Baird (2011).

Precipitation and traffic data were used together with the results of the rainfall simulation experiments to provide an estimate of the mass of sediment generated at each road crossing along the 7-km study section of the QC Main. A bulk density of 1700 kg/m³ (McNabb 1994) was used to estimate the volume of sediment from the calculated mass of sediment eroded. These volumes were then compared to those estimated using the WQEE. Precipitation data was collected using a Rainwise rain gauge (Rainwise Inc., Bar Harbour, Maine) at kilometre 10. Traffic data was collected by a TrafX vehicle counter (TRAFx Research Ltd., Canmore, Alta.) at kilometre 3 on the QC Main from August 2009 to July 2010. Precipitation events were defined as periods with at least 2.5 mm of precipitation, separated by at least 12 hours of no precipitation.

Turbidity monitoring

Turbidity was measured above and below two crossings from August 2009 to July 2010 (crossings 16 and 28 in Figure 2) to isolate sediment input from the road. Turbidity was measured continuously using Analite 9500 probes (McVan Instruments, Mulgrave, Victoria, Australia) and Hobo U12-006 data loggers (Onset, Bourne, Mass.), which recorded turbidity at 5-, 10-, or 15-minute intervals. Turbidity probes were calibrated using 400 and 1000 NTU standards and were checked during site visits against a portable turbidity probe to monitor potential drift in the readings. Turbidity was converted to sediment concentration using a relationship developed for sediment standards created with sediment from the watershed ($r^2 = 0.99$, $p < 0.001$). To determine the sediment flux, concentrations were combined with discharge data based on an area discharge relationship (Baird 2011). Equipment malfunctions, common in remote areas, resulted in an incomplete data set.

Results

Water Quality Effectiveness Evaluation Results

The road surface in the QC Main had a high proportion of fine sediment composed of silts and clays that are easily erodible and do not readily fall from suspension. The QC main was classified as heavily used, which increases the availability of erodible material. Road surface slopes ranged from 0 to 10%, and averaged 2%. The WQEE estimated a total erosion potential of 235 m³/yr for the 7-km study section of QC Main, with 74% (175 m³/yr) of the eroded sediment predicted to reach the Honna River. The sediment generation ratings of the 52 crossings were:

- 18 very low or low (35%),
- 22 moderate (42%),
- 12 high (23%), and
- 0 very high (Figure 4).

The majority of crossings rated as moderate and high were located near kilometre 8 of the QC Main (Figure 2), where poor quality road surface material (easily erodible sandstone and siltstone) is present. The river is also close (3–10 m) to the road near kilometre 8, resulting in high connectivity.

The WQEE survey identified the road surface as the major source of road-generated sediment along the QC Main. Ditches, cutslopes, and fillslopes were generally well vegetated and riprap was present in various locations to prevent cutslope erosion. Culverts along the QC Main were frequent and well spaced. Several ditches contained ditch blocks, at times covered in geotextile material.

### Table 1. Rating of total fine sediment generation (independent of stream size)

<table>
<thead>
<tr>
<th>Total volume of fine sediment generated</th>
<th>Site sediment generation potential classes</th>
<th>General level of management</th>
</tr>
</thead>
<tbody>
<tr>
<td>&lt; 0.2 m³</td>
<td>Very low</td>
<td>Good</td>
</tr>
<tr>
<td>0.2–1 m³</td>
<td>Low</td>
<td></td>
</tr>
<tr>
<td>1–5 m³</td>
<td>Moderate</td>
<td></td>
</tr>
<tr>
<td>5–20 m³</td>
<td>High</td>
<td></td>
</tr>
<tr>
<td>&gt; 20 m³</td>
<td>Very High</td>
<td>Poor</td>
</tr>
</tbody>
</table>

*This table is derived from Table 8 in Carson et al. (2009:33).*

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to promote pooling of water and to decrease the amount of ditch flow reaching the river. The effectiveness of the ditch blocks was limited because these filled soon after ditch flow was initiated, letting sediment-laden water pass through, often reaching the Honna River (Figure 5).

**Rainfall Simulator**

Rainfall simulation results were used to develop a regression equation that gave the mass of sediment produced from the area below the simulator during a precipitation event based on rainfall intensity and road use conditions:

\[
M = 723I + 512n
\]

adjusted \( r^2 = 0.85 \) [1]

where \( M \) is the mass of sediment generated by the event in grams; \( I \) is the peak precipitation intensity in millimetres per hour; and \( n \) is the number of loaded logging trucks passing during the event. Adding rainfall amount and truck speed did not significantly improve the regression. Applying the 5% and 95% confidence limits of this regression equation to the road surfaces at each crossing between the kilometre 3 and 10 points of the QC Main, along with precipitation and traffic data, the estimated sediment generated between August 2009 and July 2010 was \( 3.4 \times 10^1 \) to \( 5.8 \times 10^2 \) m\(^3\)/yr. This is in general agreement with the results of the WQEE assessment (Table 2). The unit area estimates of sediment generation were \( 5.8 \times 10^3 \) m\(^3\)/km\(^2\) per year for the WQEE assessment, and \( 6.5 \times 10^3 \) m\(^3\)/km\(^2\) per year (\( 8.4 \times 10^2 \) to \( 1.4 \times 10^4 \) m\(^3\)/km\(^2\) at the 95% confidence interval) for the rainfall simulation method. Compared to the rainfall simulation, the WQEE frequently underestimated the sediment volumes generated from the road surface at low sediment-producing crossings and overestimated those at high-producing crossings (Figure 6).

**Turbidity Above and Below Crossings**

Turbidity was consistently higher below stream crossings than above stream crossings (Figure 7). Small tributaries had rapid, frequent, and large turbidity responses, even during small
rainfall events (Figure 8). Equipment malfunctions caused the turbidity and discharge data to be insufficient to determine an annual sediment flux to compare to the WQEE results for crossings 16 and 28; however, the data from these sites suggest that on an annual basis the high rating of the WQEE at crossing 16 and moderate rating at crossing 28 are appropriate.

**Discussion**

Total sediment yields estimated by the WQEE assessment and those derived from the rainfall simulation were comparable for the QC Main in the Honna River watershed. Nevertheless, the WQEE estimates of sediment volumes generated from the road surface were lower at low sediment-producing crossings and higher at high-producing crossings (Figure 6).

Figure 6. Comparison of WQEE and rainfall simulation (RS) based yearly sediment yield estimates.

Figure 7. Monthly percent time turbidity values were exceeded above and below crossings 16 (high rating) and 28 (moderate rating).
The rainfall simulation method does not incorporate any spatial variation in road surface material, bulk density, or slope, which may explain this deviation. Even with these potential sources of error, the amount of sediment generated from the road surface based on rainfall simulation results was 10% more compared to the WQEE results, which is well within the expected error based on the WQEE rating classes.

Even with the potential uncertainty in calculating volumes using the WQEE at individual locations, this method provided an efficient means to assess multiple crossings. One field technician with minimal training took 2 days to complete the WQEE assessment of 52 crossings along the QC Main. Placing instruments above and below a road crossing can cost up to $15,000 and requires installation, maintenance, and data processing, thus limiting its application. Even in very controlled situations, it is difficult to acquire good quality data for long periods. However, direct measurements of stream sediment can be useful in evaluating the performance of methods similar to the WQEE. Unlike large field measurement campaigns, which are expensive and subject to annual fluctuations in weather, assessment procedures like the WQEE can be repeated through time to show trends in resource road sediment hazard or risk. However, field-based procedures like the WQEE should be evaluated with experimental data to gain confidence in the assessments and then be modified as needed to ensure accurate results. Combining field measurements designed to investigate erosion processes, such as the rainfall simulations, with turbidity measurements is a unique way to advance our scientific understanding of erosion processes and evaluate assessment procedures used for management. Indicator-based road assessment procedures will likely continue to be a cost-effective way to assess these roads. Equally important is to determine whether repairing sections of roads or individual crossings will improve water quality at both the site and watershed scale.

As part of the greater research project in the Honna watershed, turbidity probes were installed along the length of the river to determine whether road-related sediment from the QC Main had a measurable effect on water quality at the drinking water intake. To date, 1 year of data suggests that the QC main contributes between 5% and 35% of annual sediment yield to the Honna River and that small tributaries frequently lack sufficient discharge to dilute sediment, which possibly enters from ditches or road crossings (Baird 2011). These amounts are significant because of the relatively small area of road compared to the total area of the watershed; therefore, improving the most problematic road sections may substantially improve water quality. In phase 2 of this project, which is expected to be completed in 2014, two additional years of turbidity data will be assessed to determine the causes of increased turbidity in the Honna River, with a focus on vehicle traffic and chronic sediment inputs not associated with large precipitation events. In addition, sections of the QC Main identified as sources of sediment
will be repaired and additional rainfall simulation experiments completed to determine whether sediment delivery from the road has been reduced.

Summary

Although the WQEE results and rainfall simulation estimates of the annual volumes of sediment generated by the QC Main were similar, some discrepancies were evident in the volumes calculated at individual crossings. It is not surprising that both methods produced different volume estimates for some crossings because of the variability in road conditions. Turbidity monitoring above and below stream crossings indicated that the WQEE classified the impacts on these crossings appropriately, but there was insufficient turbidity data for a complete annual evaluation. Although such measurements and experiments are capital- and time-intensive, they can help to evaluate the results of an indicator-based assessment, which is designed for use by non-specialists, thereby increasing confidence in its application and providing insights for future improvements.

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Introduction

Elevated delivery of fine-grained sediments into streams can adversely affect downstream water quality. Effects of fine sediment include direct and indirect impacts on fish, invertebrates, and aquatic plants, as well as diminished potability, recreational value, and aesthetics of the affected watercourse (Kerr 1995). Land use activities can elevate sediment yields from forested catchments by increasing erosion rates on cleared slopes, initiating road surface erosion, and increasing delivery of sediment to stream channels by debris slides from timber harvest areas (Church 2010). Relatively few studies have directly examined the impacts of land use on lacustrine (lake) sedimentation rates in the Canadian Cordillera, with the notable exceptions of research conducted in British Columbia by Arnaud and Church (1997), Spicer (1999), and Schiefer et al. (2001a). These studies show increased lake sedimentation with timber harvesting and road construction activities. Signatures of land use may be confounded, however, by natural disturbances and the complex response of the catchment system.

All three of the aforementioned studies used \(^{210}\text{Pb}\) radiometric dating of lake sediment deposits to produce accumulation rate profiles for the last 100–150 years. These profiles of sediment accumulation rates were then compared with historical information about natural and anthropogenic disturbances in contributing catchments since the early to mid-20th century. In their detailed study of four coastal lake catchments, Arnaud and Church (1997) observed significant increases in lake sedimentation coinciding with forestry-related disturbances, natural disturbances such as extreme rainstorms, and other anthropogenic activities such as mining. In a study of 11 central interior and 10 coastal catchments, Spicer (1999) found that the onset of forestry disturbances, wildfire activity, and major earthquakes and storms could be related to increased sedimentation, with the proximity of forestry disturbances to stream channels and catchment slope characteristics influencing the severity of land use impacts on sedimentation rate. In a study of 70 headwater lake catchments throughout northwestern British Columbia, Schiefer et al. (2001a) observed regionally variable trends in sedimentation and increasing sedimentation rates irrespective of land use change. This temporal trend in sedimentation may have been related to precipitation increases during the late 20th century. Storms and other episodic natural disturbances often appeared to dominate sediment transfer, particularly in the montane subregions of that study. In northwestern British Columbia, land use impacts could only be partially separated from natural processes; however, the signature of land use change was observed in some of the study catchments that were exposed to particularly high levels of land use disturbance.

The impacts of land use on sediment transfer through watersheds and associated lacustrine sedimentation have not been systematically investigated within or east of the Rocky Mountains. Here, we present a new lake sedimentation database for the Rocky Mountain foothills of west-central Alberta, which was obtained to further examine regional land use impacts across the Canadian Cordillera. This eastern region of the Cordillera is of special interest because of its unique land use history and physiographic setting. This article presents preliminary results on the relationship between land use and lacustrine sedimentation, and proposes a new longitudinal study design for re-analyzing all of the available lake sedimentation databases for western Canada.

Study Area

The Rocky Mountain Foothills region represents a transitional climatic environment of western Canada, with cold winters typical of the boreal forest regions and high winter snowfall typical of the western mountain ranges (Natural Regions Committee 2006). Daily average temperatures range from close to 15°C in July to about –12°C in January, and average total annual precipitation exceeds 550 mm with about 175 cm of annual snowfall (Environment Canada 2011). Peak monthly precipitation, primarily resulting from convective activity, occurs during the summer months of June and July. Relative to other cordilleran climates, the foothills region is most influenced by continental air masses and it receives the most heavily modified Pacific weather systems.
The study area spans the Southern Rocky Mountain Foothills and the adjacent Alberta Plateau physiographic regions of western Canada (Mathews 1986) and the Lower to Upper Foothills and Central Mixwood natural regions of west-central Alberta near Edson (Figure 1). Surficial materials are dominantly a medium-textured till blanket and underlying bedrock is comprised of Paleocene sedimentary strata of the Paskapoo Formation (Alberta Geological Survey 2011). Gently undulating hills and plateaus with deciduous and mixed-wood forests characterize the region in the east and more steeply rolling hillslopes with coniferous forests in the west. Forest cover is mostly continuous except where water features exist or where areas have been cleared by land use activities.

Standing water bodies in the region are dominated by wetland features. Although the lakes are relatively small in size, they are important sites for recreation, including camping, boating, fishing, and swimming. Land use in contributing catchment areas is dominated by timber harvesting and energy resource extraction. Over 50 years of forestry activities, combined with intensive oil and gas exploration and extraction over the past several decades, have resulted in a high degree of landscape fragmentation with the development of a dense network of roads, cutblocks, seismic cutlines, wells, and pipelines throughout the region (Figure 2). Urban development is minimal and agricultural activity is mostly restricted to the southeastern fringe of the study area.

Methods
Thirteen lakes were selected for the study in the foothills region surrounding Edson, Alberta (Figure 1). Study lakes were selected on the basis of three criteria:

1. contributing catchment areas exhibited a range of historic land use intensities;
2. lakes had a relatively simple bathymetry; and
3. lakes were deep enough (> 5 m) to minimize the effects of wind mixing, river currents, and bioturbation of bottom sediments.

Figure 1. Locations of study lakes plotted over natural regions of Alberta (Natural Regions Committee 2006).

Figure 2. Sample area of the Lower Rocky Mountain Foothills northwest of Edson, Alberta, with high forestry- and energy-related land use intensities. Visible disturbances include cutblocks, roads, seismic survey cutlines, and natural gas well sites. Width of area shown is approximately 5 km (2011 SPOT imagery).
Sediment cores were extracted from the deepest part of each lake using a 7.6 cm diameter Kaja-Brinkhurst gravity corer (Glew et al. 2001) during the summer of 2008. The average length of the sediment cores was 34 cm (range: 21–44 cm). Cores were extruded lake-side at a 0.5-cm interval and the approximately 22-cm³ (wet) subsamples were submitted to MyCore Scientific Inc. for 210Pb dating using a constant rate of supply model (Turner and Delorme 1996).

Land use characteristics were measured from a digital watershed inventory developed for the 13 lake catchments using several geospatial data sources and GIS software. Topographic data was obtained from the National Topographic System database for Canada (Natural Resources Canada 2009). Land use features were dated and digitized from the available air photo record housed at the provincial Air Photo Reference Library in Edmonton, Alberta. For most of the study catchments, air photos were available since the mid-20th century with a repeat coverage interval of approximately one decade. Air photos were mostly panchromatic with nominal spatial scales ranging from 1:15 000 to 1:60 000. To capture and date more recent land use features in the absence of recent air photos (post-2000), other remotely sensed imagery was used (e.g., Landsat).

Suites of landscape (static) and land use (dynamic) indices were extracted from the watershed inventories using GIS scripts. Static landscape indices included catchment area, lake area, area of other water features, drainage density, and multiple slope and elevation statistics. Dynamic land use indices included spatial densities of clearcuts (km²/km²), unpaved roads (km/km²), seismic cutlines (km/km²), and number of well pads (number/km²). Immell (2011) provides a more detailed description of the study methods.

To compare rates of sedimentation with land use, averaged sedimentation rates were calculated using the time intervals for characterizing land use change from the available imagery (i.e., based on dates of air photos and other remote sensing imagery). The number of dated sediment subsamples exceeded the number of imagery dates by over a factor of two, with 16.0 dated subsamples to 6.8 imagery dates on average per lake catchment. Sedimentation rates following the onset of land use were converted to percent changes relative to pre-land use (background) rates of sedimentation.

**Tables**

<p>| Table 1. Selected landscape and land-use index ranges from the digital watershed inventory. |</p>
<table>
<thead>
<tr>
<th>Morphometric variables</th>
<th>Range</th>
<th>Land use variables</th>
<th>Range</th>
</tr>
</thead>
<tbody>
<tr>
<td>Catchment area</td>
<td>0.5–273 km²</td>
<td>Proportion harvested</td>
<td>0–0.41 km²/km²</td>
</tr>
<tr>
<td>Lake elevation</td>
<td>773–1375 m</td>
<td>Road density</td>
<td>0–2.39 km/km²</td>
</tr>
<tr>
<td>Drainage density</td>
<td>0–0.86 km/km²</td>
<td>Cutline density</td>
<td>0–3.4 km/km²</td>
</tr>
<tr>
<td>Mean slope</td>
<td>0.2–9.4°</td>
<td>Well density</td>
<td>0–2.7 wells/km²</td>
</tr>
</tbody>
</table>

<p>| Table 2. Regression results relating sedimentation rate increase with land use indices. |</p>
<table>
<thead>
<tr>
<th>Land use variable</th>
<th>Distance from water features</th>
<th>Intercept (SE)</th>
<th>Slope (SE)</th>
<th>Adjusted $R^2$</th>
<th>p-value</th>
</tr>
</thead>
<tbody>
<tr>
<td>Roads (km/km²)</td>
<td>&lt; 10 m</td>
<td>38 (22)</td>
<td>147 (54)</td>
<td>0.35</td>
<td>0.020</td>
</tr>
<tr>
<td></td>
<td>&lt; 50 m</td>
<td>33 (19)</td>
<td>98 (28)</td>
<td>0.49</td>
<td>0.005</td>
</tr>
<tr>
<td></td>
<td>&lt; 250 m</td>
<td>71 (31)</td>
<td>7 (21)</td>
<td>0.08</td>
<td>0.747</td>
</tr>
<tr>
<td></td>
<td>&lt; 500 m</td>
<td>82 (30)</td>
<td>–2 (14)</td>
<td>0.09</td>
<td>0.900</td>
</tr>
<tr>
<td>Wells (number/km²)</td>
<td>&lt; 10 m</td>
<td>76 (22)</td>
<td>32 (88)</td>
<td>0.08</td>
<td>0.719</td>
</tr>
<tr>
<td></td>
<td>&lt; 50 m</td>
<td>50 (16)</td>
<td>216 (63)</td>
<td>0.47</td>
<td>0.006</td>
</tr>
<tr>
<td></td>
<td>&lt; 250 m</td>
<td>51 (21)</td>
<td>139 (63)</td>
<td>0.24</td>
<td>0.050</td>
</tr>
<tr>
<td></td>
<td>&lt; 500 m</td>
<td>52 (22)</td>
<td>141 (68)</td>
<td>0.22</td>
<td>0.061</td>
</tr>
</tbody>
</table>

**Results**

The lake sediment samples were dominantly massive (i.e., lacking visible sediment structure) and ranged in colour between the gray, black, and olive-yellow fields of the Munsell colour system. Background sediment accumulation rates, determined by 210Pb dating, ranged from 36 to 284 g/m² per year for the study lakes. Mean increases in sedimentation rates following the initiation of land use activities ranged from about 0 to 250%. Ranges of some primary landscape and land use indices are listed in Table 1. Catchments in the western portion of the study area tend to have been more dominated by forestry-related land use activity (e.g., Musreau Lake; Figure 3a), whereas catchments in the east tend to have been more dominated by energy-related land use activity (e.g., Fickle Lake; Figure 3b). Immell (2011) provides a complete tabulation and summarization of the data results.
Preliminary analyses revealed no strong correlations between estimated background rates of sediment yield and morphometric variables for the lake catchments. Moderate correlations were observed, however, between total road and well densities in proximity to watercourses and sedimentation rate increases following land use activities (Table 2). Strongest bivariate relations were observed for total road and well densities within 50 m of water features (Figure 4). A moderately strong relation was observed between sedimentation increases and both road and well densities together in a multivariate regression model; however, this model was not statistically more significant than those that only considered roads or wells individually. This may largely reflect the high degree of collinearity between road and well densities ($r = 0.70$). The weak relation between wells highly proximal to watercourses ($< 10$ m) and sedimentation rate increases is likely spurious because of the low total well count within that buffer distance ($< 1$ on average per lake catchment). No significant relations were observed between the other two land use variables (clearcut area and seismic line density) and sedimentation rate increases in this analysis. We did not attempt to relate sedimentation rate increases with climate change variables because regional precipitation intensities have generally decreased since the mid-20th century and a clear signal of regional warming is only evident in the meteorological records for the last two decades.

**Discussion**

The range of background sedimentation rates observed in this study closely matches the range of background sedimentation rates observed by Schiefer et al. (2001a) for the Nechako Plateau in northwestern British Columbia. Both regions are similar in their relief characteristics and both are underlain by sedimentary bedrock strata. Unlike the Nechako Plateau...
region, where lower specific sediment yield estimates were associated with larger catchment areas because of increased sediment trapping potential downstream in larger drainage basins (Schiefer et al. 2001b), background specific sediment yields in this study were not well predicted by any linear combination of morphometric variables describing the catchment areas. Most landscape characteristics are relatively consistent across the study region, but catchment area does vary by over two orders of magnitude. In the Rocky Mountain Foothills and adjacent Alberta Plateau, sediment yield appears to increase downstream in simple proportion to the area drained. In much of western Canada, sediment yields tend to increase out of proportion to drainage areas because the remobilization of Quaternary sediment deposits increasingly dominates fluvial sediment sources downstream (Church et al. 1999).

With the overlap of both forestry and energy resource extraction throughout much of the foothills region, land use intensities in the study catchments are high relative to other regions of the Canadian Cordillera. Overall, sedimentation rates since the mid-20th century in all but one of the study lakes have increased significantly. Lakes with greater increases in sedimentation generally correspond with catchment areas that experienced the greatest intensity of land use disturbance. Our preliminary analyses showed that the total density of roads and wells in proximity to watercourses could be related to the magnitude of sedimentation rate increases since the onset of land use.

To a large extent, land use effects on lake sedimentation may have been difficult to discern in our preliminary analysis and in the analyses of other lake sedimentation data sets from elsewhere in the Cordillera (i.e., Arnaud and Church 1997; Spicer 1999; Schiefer et al. 2001a) because of the large number of confounding variables that can be associated with sediment transfer. Such confounding variables include relatively static landscape variables (e.g., catchment morphometry) and dynamic or time-varying variables (e.g., land use and climate variables). In addition, there is also likely a significant amount of unexplained or unknown sources of catchment-specific variability that we cannot deterministically model because of the high degree of complexity in sediment transfer processes, both spatially and temporally, at the catchment scale. To account for these issues, we are currently developing what we anticipate will be a more appropriate modelling approach for these data sets. The approach being considered will use a mixed-effect model (Wallace and Green 2002) to explicitly separate fixed effects (i.e., variance in sedimentation associated with independent model variables) from random effects (i.e., catchment-specific effects not associated with model variables). Such a method is well suited for repeated measure data where a dependent variable (i.e., sedimentation rate) and some controlling independent variables (i.e., land use and climate variables) are observed on multiple occasions (e.g., \(^{210}\text{Pb}\) dating intervals and air photo dates) for each experimental unit (i.e., the lake catchment). This kind of analysis can incorporate both static and time-varying covariates associated with the repeated observations, allowing for better statistical inferences of land use effects by increasing degrees of freedom beyond the limited number of lake catchments under study. Following the refinement of this new methodology, we would like to perform a retrospective analysis of all the other \(^{210}\text{Pb}\)-based lake sedimentation data sets available for the Canadian Cordillera, which now exceeds 100 lake catchments in total, to more thoroughly explore regional land use impacts on lacustrine sedimentation.

**Conclusion**

In a preliminary analysis of lake sedimentation trends in the Rocky Mountain Foothills region of west-central Alberta, increases of sedimentation rates since the mid-20th century were most positively correlated with road and well densities in relative proximity to watercourses. Weak or insignificant relations were observed between sedimentation rates and cutblock or seismic cutline densities, or morphometric variables for the lake catchments. We are currently exploring a multivariate, mixed-model approach to more robustly investigate cumulative land use effects and other climatic and morphometric controls of lacustrine sedimentation rates in this region. Anthropogenic erosion and sedimentation can be an important environmental issue for watershed management, as elevated fine-grained sediments in streams and lakes can be harmful to aquatic organisms and degrade water resources for human use. Since current hydrometric monitoring stations rarely collect any sediment transport data, the lake sediment-based approach may become increasingly important in the study of long-term patterns of sediment transfer, particularly for geomorphically dynamic and remote regions. The lake sediment approach has been successful in identifying 20th century land use impacts on sediment transfer in diverse catchment systems throughout western Canada. New study techniques may produce more robust statistical relations among landscape characteristics, climatic controls, land-use impacts, and sediment transfer to aquatic environments.
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Cumulative Effects Assessment: Runoff Generation in Snowmelt-dominated Montane and Boreal Plain Catchments

Russell Smith and Todd Redding

Introduction

Cumulative effects assessments (CEAs) have been conducted in North America for decades and have been largely motivated by regulatory requirements (e.g., Canadian Environmental Assessment Act\(^1\)) related to industrial development. Many formalized CEAs have focussed on ecological and (or) water quality impacts; however, as issues regarding water quantity become more common, the need to manage cumulative effects on water supply will become more urgent. Designing and implementing an effective CEA process for runoff (i.e., surface and [or] subsurface drainage of water from an upslope area to a stream or open water body) will help manage impacts to source water supply.

The surface water supply in much of British Columbia and Alberta is generated primarily by snowmelt in forested catchments within mountainous and (or) plateau areas (e.g., Peace, Athabasca, North Saskatchewan, Fraser, and Columbia rivers). How, where, and when runoff is generated from hillslopes and translated into streamflow in forested catchments is important to understand, as runoff generation processes control streamflow response to rainfall and snowmelt, and runoff rates and flowpaths affect water quality and the initiation of mass movements.

This article provides an overview of runoff generation processes and related CEA tools as a resource for land managers and policy decision makers in designing and implementing a CEA of runoff in snowmelt-dominated catchments. Specifically, it:

1. briefly reviews how runoff is generated and what environmental factors control runoff dynamics with an emphasis on contrasting boreal plain and snowmelt-dominated montane catchments;
2. identifies indicators of the sensitivity of runoff generation to disturbance and indicators of the potential for changes to runoff regimes;
3. briefly outlines office- and field-based approaches for evaluating indicators with a summary of some potential data sources and available tools; and
4. identifies key gaps in knowledge.

Controls on Runoff Generation Dynamics

Runoff generation is the development of lateral water movement on or below the soil surface. Three different spatial scales are important in understanding runoff generation dynamics: (1) plot, (2) hillslope, and (3) catchment scales.

The plot scale incorporates runoff processes occurring in the vertical direction (i.e., between the atmosphere, vegetation, soil, subsoil [material below the soil and above the bedrock], and groundwater) and over short horizontal distances (e.g., maximum 5–10 m). The hillslope scale focusses on vertical and lateral runoff processes occurring along a relatively uniform hillslope section spanning the distance from a ridge top to the bottom of its neighbouring hollow or valley (e.g., from ~50 m to 1000 m or more). The catchment scale focusses on runoff from multiple hillslopes and (or) tributary catchments converging into one main catchment. It typically incorporates a greater range of climatic, vegetative, soil, and topographic conditions compared to the plot and hillslope scales.

Water infiltrating the soil surface tends to move vertically downward (i.e., percolation) due to the effects of gravity and capillary forces (i.e., tension in the soil pores), unless restricted by a soil or subsoil layer or bedrock with a permeability that is lower than the rate of percolation, or by the presence of a water table. The depth to the restricting layer is a function of the rate of water input or percolation relative to the capacity for vertical flow within the soil or subsoil. For example, if the water table is at the soil surface, saturation overland flow will occur. Where the rate of water input at the soil surface is greater than the infiltration capacity of

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the soil, infiltration-excess overland flow will occur. If the water is able to infiltrate and a permeability contrast exists within the soil profile (e.g., soil layering or bedrock/compacted till surface) creating a restricting layer, percolating water will be re-directed laterally as subsurface flow (Hutchinson and Moore 2000; Redding and Devito 2010; Smith 2011). If the soil has a relatively homogeneous pore structure (i.e., minimal interconnected macropores creating preferential flow pathways), water will move in the steepest downslope direction of the restricting layer. In the absence of a restricting layer, water that does not become stored within the unsaturated soil will continue to percolate until it reaches the regional water table and results in groundwater recharge (Redding and Devito 2010; Smith 2011). A conceptual model of the threshold controls on vertical and lateral flow for luvisolic soils at a site on the boreal plain in Alberta is provided in Figure 1 (adapted from Redding and Devito [2010]); however, the specific thresholds are site specific and depend on the storage and transmission properties of the soil, subsoil, and bedrock (Weiler et al. 2005; Smith 2011).

While lateral flow develops at the plot scale, hillslope hydrologic connectivity, which varies in space and time, strongly influences the delivery of runoff to the stream network. For instance, water flowing downslope can be disconnected from the stream network (i.e., not contribute to streamflow) by contributing to water storage within unsaturated soils in the downslope areas (McNamara et al. 2005; Kuras et al. 2008; Jencso et al. 2009). A similar flowpath disconnect can occur where the percolation-restricting layer is discontinuous, as water flowing downslope can begin percolating to deeper unsaturated soils in locations where the restricting layer is absent. When a contiguous section of lateral flow connects most or all of the hillslope to the stream network, water can be rapidly delivered for streamflow (Dunne 1983; Sidle et al. 2000; Jencso et al. 2009). As a catchment is the accumulation of multiple individual hillslopes, the dynamics of runoff contributions to the stream network are controlled by the complex integration of hillslope-scale runoff processes occurring throughout the catchment (Smith 2011).

Several factors influence the space-time distribution of runoff generation and hillslope hydrologic connectivity. The soil wetness at the start of a water input event (i.e., antecedent wetness) strongly influences the lateral flow response, as any storage of infiltrating water reduces the amount of water available for percolation and (or) lateral flow (Sidle et al. 2000; Grant et al. 2004; Kim et al. 2005). The depth to the percolation-restricting layer influences the soil water storage capacity, so deeper soils require more water to satisfy storage before lateral flow is generated (Redding and Devito 2008, 2010). The quantity and timing of water inputs to the soil surface relative to that of evapotranspiration (ET) demand also influence antecedent soil wetness. Where a percolation-restricting layer exists within the shallow soil, topographic convergence (i.e., concave/convergent topography versus convex/divergent topography) controls the accumulation of soil water and, thus, soil wetness and associated lateral flow responsiveness (Thompson and Moore 1996; Sidle et al. 2000; Smith 2011). Table 1 provides a summary of the factors influencing the space-time distribution of runoff generation processes.

Catchment-scale runoff generation processes are complex, particularly in snowmelt-dominated areas. Large space-time variability of water inputs

Continued on page 26
(Toews and Gluns 1986; Winkler et al. 2005) and corresponding generation of lateral flow combined with variability in hillslope hydrologic connectivity creates potential for synchronization and (or) desynchronization of runoff contributions throughout the catchment (Deng et al. 1994; Kuras et al. 2008; Smith 2011). The following scenarios illustrate this complexity.

- Scenario 1: Two nearby hillslopes in a given catchment that receive water inputs to the soil surface at the same time might contribute to streamflow at different times and rates because of differences in antecedent soil wetness, soil depth, soil permeability, vegetation cover, and (or) topographic convergence.

- Scenario 2: If the two hillslopes have opposite slope aspects (e.g., north-facing versus south-facing) and are within a snowmelt-dominated catchment, it is plausible that they might receive water inputs weeks or months apart because of variable snowmelt timing (caused by variable meteorological conditions) leading to runoff contributions to the stream network that are highly desynchronized in time.

**Differences in Controls on Runoff Generation Between Snowmelt-dominated Montane and Boreal Plain Catchments**

The principles governing runoff generation discussed in the previous section apply to both snowmelt-dominated montane and boreal plain catchments; however, differences in climatic conditions and the spatial distributions of vegetative, soil, geological, and topographic properties result in varied hydrologic responses to water inputs. In snowmelt-dominated montane catchments throughout British Columbia and Alberta, most of the water inputs to the soil occur during the spring snowmelt period when evapotranspiration rates are low, producing a surplus of water available for streamflow or groundwater recharge.

### Table 1. Factors influencing the space-time distribution of runoff generation processes. Abbreviations: \( P \) (precipitation), \( ET \) (actual evapotranspiration), and \( PET \) (potential evapotranspiration).

<table>
<thead>
<tr>
<th>Factor</th>
<th>Details</th>
</tr>
</thead>
<tbody>
<tr>
<td>( P-ET ) (or ( P-PET ))</td>
<td>Represents an annual and (or) seasonal assessment of the water balance and an estimate of the depth of water available for runoff, storage, or groundwater recharge. Influences antecedent soil wetness. Runoff is most common during periods when ( P &gt; ET ).</td>
</tr>
<tr>
<td>Vegetation cover</td>
<td>Influences the quantity of ET and, thus, antecedent wetness. Influences interception of precipitation, which varies seasonally for deciduous vegetation (higher in summer than winter). Influences shading of the snow surface, which affects energy exchange between the snowpack and atmosphere and, thus, the accumulation of snow and the quantity, timing, and rates of snowmelt.</td>
</tr>
<tr>
<td>Seasonal distribution of water inputs</td>
<td>Influences the extent to which water inputs are concentrated in time and how the timing relates to periods of high or low ET, which affects the potential for runoff. Influenced by the seasonal distributions of precipitation and air temperature, as they control snowpack processes and the occurrence of seasonally high rainfall.</td>
</tr>
<tr>
<td>Organic soil depth</td>
<td>Influences the retention of water above the mineral soil, which can contribute to plant water uptake.</td>
</tr>
<tr>
<td>Mineral soil depth</td>
<td>Influences the soil water storage capacity and the depth of percolation to a restricting layer. Influences the attenuation of lateral flow.</td>
</tr>
<tr>
<td>Soil and subsoil permeability</td>
<td>Influences the infiltration capacity, which can control the occurrence of infiltration-excess overland flow. Influences water ponding on the soil surface. Influences the occurrence and depth of the percolation-restricting layer and, thus, the potential for saturation overland flow, lateral flow in the soil, percolation into the subsoil, and groundwater recharge. Forest soils commonly contain macropores, which promote rapid vertical and lateral flow via preferential flow pathways. As a result, fine-textured soils, which would otherwise have low permeability, can experience rapid lateral flow. The tendency for macropore flow can be positively related to the presence of vegetation with extensive root structures, the abundance of burrowing animals, and the proportion of clay in the soil (due to the shrink/swell behaviour of many types of clay).</td>
</tr>
<tr>
<td>Bedrock type and topography</td>
<td>Influences the permeability of bedrock, which can control whether percolating water flows vertically or laterally. Influences flow pathways at both large (e.g., regional) and small (sub-catchment) scales. Bedrock topography can control flowpaths at the bedrock-subsoil interface.</td>
</tr>
<tr>
<td>Slope aspect and gradient</td>
<td>Influences the intensity of incoming solar radiation, which controls the quantity of ET and antecedent wetness, and the timing and rates of snowmelt. Influences the rate of lateral flow towards the stream network via the influence of gravity.</td>
</tr>
<tr>
<td>Topographic convergence</td>
<td>Influences the accumulation of water on the hillslope and, thus, the rates and magnitude of water contribution to the stream network.</td>
</tr>
</tbody>
</table>
and a distinct spring freshet period. Across most of the boreal plain, annual precipitation is equal to or less than annual ET and most water inputs occur in June, July, and August, which coincides with the timing of high ET rates and results in little surplus water for runoff in most years. Snowmelt-generated streamflow is low in most years, as water inputs must satisfy the soil water deficit from the preceding fall before surplus water is available for streamflow or groundwater recharge (Devito et al. 2005a; Redding and Devito 2011). Unlike montane catchments, annual peak flows on the boreal plain are most commonly driven by summer rain events rather than spring snowmelt (Redding and Devito 2011).

In many montane catchments, soils are relatively shallow (< 1–2 m), highly permeable, and are underlain by relatively impermeable bedrock or glacial till. Shallow soils increase runoff responsiveness in several ways (compared to deep soils):

- by limiting the soil water storage capacity and, thus, the influence of soil water storage on runoff attenuation;
- by increasing the potential for transient soil saturation at or near the soil surface, which increases the potential for saturation overland flow and preferential flow; and
- by limiting deep percolation and regional groundwater recharge.

Moreover, topographic variability tends to strongly control the distribution of rapid lateral flow in catchments with shallow soils during large water input events and during periods with high antecedent soil wetness. Although these points broadly represent the conditions in many montane catchments (Thompson and Moore 1996; Sidle et al. 2000), there are exceptions. For instance, Smith (2011) observed highly permeable soils exceeding 8 m in depth in the Cotton Creek Experimental Watershed in southeast British Columbia. Locations with deep, coarse soils experienced little or no runoff generation, and the distributions of deep soil permeability and water input timing and intensity exerted greater control than topography on the distribution of runoff during early spring freshet periods. Moreover, researchers in other montane catchments found that bedrock permeability exerted a significant control on runoff flowpaths and rates (Montgomery et al. 1997; Montgomery et al. 2002; Tromp-van Meerveld et al. 2007).

In contrast to many montane systems, the boreal plain is a mosaic of shallow and deep (> 20 m; Devito et al. 2005c) glacially derived soils and subsoils (till, outwash sands and gravels, lacustrine clays, and silts), and wetlands. The relative distributions and properties of the materials are largely responsible for controlling the dominant flow pathways and interactions between groundwater and surface water (Devito et al. 2005c).

The distribution of soil depths coupled with a precipitation deficit (i.e., P < PET) results in a patchwork of areas experiencing infrequent shallow lateral flow separated by areas dominated by deep percolation. Under these conditions, topography exerts limited influence on the distribution of runoff generation (Devito et al. 2005c).

Analysis by Devito et al. (2005a) for the Lac La Biche area of the boreal plain in central Alberta indicated that significant hillslope contributions to regional streamflow occur approximately once every 20 years and are strongly controlled by the timing and sequence of wet and dry years, as most streamflow in this area is generated by rising groundwater levels in wetlands. Large connected wetlands, such as fens and open water bodies, tend to be the conduits of surficial flow throughout the boreal plain, particularly in low relief areas, creating a direct connection between groundwater levels and runoff at the catchment scale (Devito et al. 2005c).

In areas dominated by sandy outwash deposits, the regional groundwater table controls the elevations of local groundwater tables and open water bodies such as ponds and lakes due to the high permeability of the soils (Devito et al. 2005c; Smerdon et al. 2005). These areas are dominated by vertical flow and groundwater recharge, and the amount of recharge is related to the depth to the water table (i.e., soil water storage capacity) and the atmospheric water fluxes (i.e., P – PET) (Smerdon et al. 2008) (Figure 2). In areas that are underlain by fine-textured till, vertical flow dominates under dry conditions (Whitson et al. 2004; Devito et al. 2005a; Redding and Devito 2008, 2010) and local groundwater can be perched close to the soil surface and disconnected from regional groundwater flow systems (Riddell 2008). However, under wet conditions, lateral flow can occur if high intensity water inputs (e.g., greater than 20 mm of storm precipitation at a 20-year return period intensity; see Figure 1) exceed the soil water storage capacity (Devito et al. 2005a; Redding and Devito 2008, 2010), which is generally low during these conditions owing to high groundwater levels.

In montane catchments, wetlands exist on benches and valley bottoms, and, thus, tend to store water received from the uplands and attenuate streamflow (Jencso et al. 2010). On the boreal plain, wetlands may occur at topographic high points because of the presence of a low permeability layer underlying the wetland but overlying higher permeable glacial deposits. These features can contribute runoff to the surrounding hillslopes when the water table in the perched wetland exceeds the elevation of the restricting layer (Riddell 2008). Another feature that is unique (or more common) to the boreal plain is the formation of concrete frost (impermeable frozen layer near the soil surface), which may lead to the generation of overland flow (Redding and Devito 2011). Although the weather conditions required to generate concrete frost occur frequently, the occurrence of frost on hillslopes can be patchy; thus, whole-hillslope connectivity caused primarily by frost is unlikely in most years (Redding and Devito 2011).

Continued on page 28
Indicators of the Sensitivity of Runoff Generation to Disturbance

Many industrial activities on the forest land base are capable of affecting the quantity, timing, or rates of runoff generation and, thus, streamflow and water supply. Activities influencing runoff generation include those that alter the quantity, timing, or rates of water inputs to the soil surface or of ET; those that alter the depth, density, or permeability of the soil; and those that can disrupt the direction of runoff flowpaths. Cumulative effects of disturbance on streamflow result from an integration of environmental sensitivities to localized activities at multiple sites across the landscape but can be magnified by the effects of scale-related non-linear response behaviour (i.e., disturbance impacts can multiply as they accumulate over increasing spatial scales). Designing and implementing an effective CEA process for runoff requires understanding runoff generation processes and being able to assess indicators of sensitivity of runoff generation to disturbance at small scales, along with understanding and assessing scale-related effects. This section focuses on localized (i.e., plot or hillslope scale) influences on runoff generation processes and associated sensitivity to disturbance, whereas the following section (Indicators of the Potential for Changes to Runoff Regimes) incorporates the influences of scale-related effects. A comprehensive CEA should consider processes and indicators at all scales.

A comprehensive list of indicators for assessing localized sensitivity to disturbance should incorporate changes to water inputs and ET. These components address the potential for changes to the water balance and to the timing of water inputs, which influence the quantity of surplus water for runoff on both annual and seasonal bases. Other important indicators incorporate the depth, texture, and permeability of the soil (including potential for concrete frost); the permeability of

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**Legend**

WT (water table), UZ (unsaturated zone), and Ks (saturated hydraulic conductivity).

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Figure 2. Conceptual model of water movement for sandy and fine-textured (silt-clay) hillslopes for the Utikuma Region Study Area in north-central Alberta (Redding 2009). Arrows indicate the direction of water flow. Thin and dashed lines indicate lower potential for flow. Inverted triangles indicate the elevation of the water table. Upflux refers to upward water movement via capillary forces and plant uptake. Abbreviations: WT (water table), UZ (unsaturated zone), and Ks (saturated hydraulic conductivity).
Weather Influences the potential development of concrete frost on stream network of disturbance. Connectivity permeability Subsoil Soil permeability Low permeability soils increase the potential for lateral flow via a reduction in the permeability of the shallow soil (Proulx and Stein 1997).

Soil depth Influences the water storage capacity of the soil and the depth to a percolation-restricting layer, which influences the potential for attenuation of lateral flow and the potential for rapid lateral flow at or near the soil surface. Surface flow has the greatest potential to be disturbed due to the potential for flowpaths and runoff rates to be altered by soil compaction. Subsurface flow has a lower potential to be altered since only activities that disturb the deeper soils (e.g., road construction) can alter flowpaths and runoff rates.

Canopy cover Influences the extent to which vegetation removal will alter ET, interception, and (or) canopy shading. The loss of canopy cover can reduce snowfall interception and, thus, increase snow accumulation. Greater snow accumulation and reduced ET can increase soil wetness and runoff. Disturbance to canopy shading affects snowmelt timing and rates, and the formation of concrete frost. The greater the reduction of canopy cover, the greater the potential for increasing runoff quantities and rates.

Vegetation community composition Wet areas of the landscape are more likely to be sources of runoff than dry areas. Mapping of vegetation communities can be used to identify wet areas and areas with a higher likelihood of runoff (e.g., fens) (Winkler and Rothwell 1983).

Texture of shallow soils Compaction of soils can alter the water storage capacity and the permeability of the upper layers of soil. Fine-textured soils have a higher potential for compaction. Can be indicated by the distribution of vegetation species and road- or trail-related soil disturbance.

Soil depth Influences the water storage capacity of the soil and the depth to a percolation-restricting layer, which influences the potential for attenuation of lateral flow and the potential for rapid lateral flow at or near the soil surface. Surface flow has the greatest potential to be disturbed due to the potential for flowpaths and runoff rates to be altered by soil compaction. Subsurface flow has a lower potential to be altered since only activities that disturb the deeper soils (e.g., road construction) can alter flowpaths and runoff rates.

Soil permeability Low permeability soils increase the potential for lateral flow by limiting vertical flow. For soils experiencing lateral flow, high permeability soils transmit runoff at higher rates than low permeability soils (unless influenced by preferential flow). Compacted shallow soils, which are commonly associated with roads, landings, and skid trails, increase the potential for overland flow.

Subsoil permeability Influences the likelihood that water percolating through the soil will generate rapid lateral flow versus continued percolation to the deep groundwater. Deep subsurface flow has a lower potential to be disturbed (compared to shallow subsurface and overland flows), as most activities cannot compact the subsoil. If disturbance does occur, it can be persistent due to the slow flow and the limited access for remediation (via natural processes or human activities).

Connectivity of disturbance activities to the stream network Greater connectivity of disturbance activities (e.g., forest cover removal, road construction, mass wasting) to the stream network results in a greater risk of disrupting flowpaths and runoff rates. Areas that are most at risk include hillside hollows in areas with percolation-restricting layers within the shallow soil, riparian areas, and floodplains.

Weather Influences the potential development of concrete frost on hillslopes and, thus, the generation of infiltration-excess overland flow via a reduction in the permeability of the shallow soil (Proulx and Stein 1997).
Table 3. Indicators of the potential for changes to runoff regimes

<table>
<thead>
<tr>
<th>Indicator</th>
<th>Details</th>
</tr>
</thead>
<tbody>
<tr>
<td>Extent of vegetation removal</td>
<td>Influences rates of ET, snowpack depth, and snowmelt rate and timing. Higher disturbance levels result in greater water inputs and wetter soils, and can increase peak flows, low flows, and annual water yields. Can be represented as equivalent clearcut area (an index of the area of forest cover removal after accounting for the impacts of stand regeneration on hydrology). Should incorporate all disturbances to the original vegetation cover (i.e., not only forestry activities).</td>
</tr>
<tr>
<td>Extent of immature vegetation</td>
<td>Influences rates of ET. Higher levels of immature vegetation result in drier soils and can decrease flows (Jones and Post 2004; Perry 2007).</td>
</tr>
<tr>
<td>Vegetation species/type</td>
<td>Influences the rate of regeneration and, thus, hydrologic recovery (e.g., aspen regenerate faster than conifers due to clonal reproduction from existing root system).</td>
</tr>
<tr>
<td>Elevation/aspect distribution of vegetation changes</td>
<td>Influences the synchronization/desynchronization of snowmelt runoff timing between different portions of the catchment. Removal of vegetation at higher elevations results in impacts to later stages of the spring freshet compared to removal of vegetation at lower elevations. Removal of vegetation on south-facing areas results in a larger advancement in the timing of snowmelt than removal on north-facing areas.</td>
</tr>
<tr>
<td>Road area or density</td>
<td>Increases rates of overland flow, intersects subsurface flows, and increases the efficiency of water delivery to the stream network. Higher areas or densities can increase peak flows. Impacts depend on local soil/geology and topography.</td>
</tr>
<tr>
<td>Road cross-drain frequency or density</td>
<td>Compensates for the potential of roads to increase rates of overland flow, intersect subsurface flows, and increase the efficiency of water delivery to the stream network by increasing rates of re-infiltration of water from roads into downslope soils instead of direct delivery to the stream network. Higher frequencies or densities can reduce the impacts of roads on peak flows.</td>
</tr>
<tr>
<td>Stream crossing density</td>
<td>Influences the efficiency of water delivery to the stream network through ditches. Higher densities can increase peak flows.</td>
</tr>
<tr>
<td>Stream density</td>
<td>Influences the potential connectivity of development activities to the stream network. Higher densities create a higher potential to alter flowpaths and runoff rates.</td>
</tr>
<tr>
<td>Area of mass wasting</td>
<td>Influences the interception of lateral flow and rapid delivery of water to the stream network, particularly for slope failures in hillslope hollows and areas with shallow soils.</td>
</tr>
<tr>
<td>Total area, distribution, and size of wetlands and open water bodies</td>
<td>In montane catchments, the potential for attenuation of runoff through storage generally increases with increasing total area, distributed extent, and size of wetlands and open water bodies.</td>
</tr>
</tbody>
</table>
areas (e.g., as a predictor of road cut depths and, thus, an indicator of the potential for intersection of subsurface lateral flow) is generally most easily completed via GIS. Hydrological models can be used to integrate catchment physiography data with spatially distributed disturbance data (natural or human caused) to generate an office-based assessment of cumulative effects.

Example evaluations of runoff-related indicators for the purpose of CEAs exist within scientific and resource management literature. For instance, the British Columbia Watershed Assessment Procedure Guidebook (B.C. Ministry of Forests 1999) outlines an approach for assigning a catchment-scale peak flow hazard (i.e., potential for increases in high flows and associated stream channel disturbance) based on calculations of equivalent clearcut area (ECA), road density, and number of stream crossings, among other factors. The British Columbia Forest Practices Board calculated equivalent clearcut area and an index of watershed sensitivity for a CEA of the Kiskatinaw River watershed in northeast British Columbia (Forest Practices Board 2011). The watershed sensitivity index was a product of seven factors addressing stream channel sensitivity, catchment topography, drainage efficiency, soil/bedrock permeability, climate type, runoff synchronization, and dominant natural disturbance type. Although some of these factors do not directly influence runoff generation, this approach represents an example of how several indicators can be combined to quantify an index for CEA purposes.

Other runoff-related tools have been used operationally in Alberta for forest management, including the ECA-Alberta model (Silins 2003), a low-complexity, lumped-parameter hydrologic model which is a numerical implementation of WRENSs (U.S. Environmental Protection Agency 1980; Swanson 1994; Swanson and Rothwell 2001). The ECA-AB model provides an estimate of changes in annual streamflow (water yield) based on the area of forest cover disturbance in a catchment, rate of forest regrowth, and water balance calculations (calculated from long-term monthly precipitation and annual streamflow data). More detailed models are also available for simulating cumulative effects on runoff generation (e.g., ForHyM, HBV-EC, UBCWM, and DHSVM); however, as model detail and complexity increase, the technical skill, time, and cost

### Table 4. Data sources in British Columbia and Alberta for calculating indicators

<table>
<thead>
<tr>
<th>Data type</th>
<th>Alberta source</th>
<th>British Columbia source</th>
</tr>
</thead>
<tbody>
<tr>
<td>Soil</td>
<td>www1.agric.gov.ab.ca/$department/deptdocs.nsf/all/sag6903</td>
<td><a href="http://www.env.gov.bc.ca/soils/">www.env.gov.bc.ca/soils/</a></td>
</tr>
<tr>
<td>Vegetation</td>
<td><a href="http://www.srd.alberta.ca/MapsPhotosPublications/Maps/ResourceDataProductCatalogue/ForestVegetationInventories.aspx">www.srd.alberta.ca/MapsPhotosPublications/Maps/ResourceDataProductCatalogue/ForestVegetationInventories.aspx</a></td>
<td>apps.gov.bc.ca/pub/dwds/home.so</td>
</tr>
<tr>
<td>Geospatial</td>
<td>xnet.env.gov.ab.ca/portal_pub/ptk</td>
<td><a href="http://www.data.gov.bc.ca/dbc/geo/index.page">www.data.gov.bc.ca/dbc/geo/index.page</a></td>
</tr>
<tr>
<td>Climate</td>
<td>climate.weatheroffice.gc.ca/climateData/canada_e.html</td>
<td><a href="http://www.genetics.forestry.ubc.ca/cfg/ClimateWNA/ClimateWNA.html">www.genetics.forestry.ubc.ca/cfg/ClimateWNA/ClimateWNA.html</a></td>
</tr>
<tr>
<td>Long-term climate and soil wetness</td>
<td>www1.agric.gov.ab.ca/$department/deptdocs.nsf/all/sag6278</td>
<td></td>
</tr>
<tr>
<td>Freshwater atlas</td>
<td>sunsite.ualberta.ca/Projects/Alberta-Lakes/characteristics5.php</td>
<td>geobc.gov.bc.ca/freshwater_atlas.html</td>
</tr>
</tbody>
</table>

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required to set up and run the model also increase. Model selection criteria and reviews of a range of hydrological models for forest management applications are available from Beckers et al. (2009) and Creed et al. (2011).

For models to perform well when applied to areas dominated by vertical water flow (i.e., where soils are deep and highly permeable), they need to represent subsurface processes well, particularly soil water storage and groundwater dynamics (e.g., HydroGeoSphere; Smerdon et al. 2008). For areas dominated by fine-textured soils and subsoils, simple threshold-based lumped models may be sufficient to simulate water cycle dynamics (Redding 2009). For simulating areas with complex topography and highly variable vegetation, models need to represent energy fluxes and snowpack processes well (e.g., distributed snow modelling at Cotton Creek Experimental Watershed; Jost et al. 2012).

Useful information and approaches for conducting CEAs related to runoff generation can be found in Devito et al. (2005b), Devito and Mendoza (2006), Pike et al. (2010), Reid (1993, 2010), Reiter and Beschta (1995), Scherer (2011), and Spafford and Devito (2005).

Gaps in Knowledge and Tools

Significant gaps exist in our knowledge and predictive tools related to runoff generation processes in snowmelt-dominated montane and boreal plain systems, including the following.

- Space-time distribution of dominant runoff generation processes, and how spatially variable soil conditions influence this distribution.
- Spatial distribution of dominant source areas for maintaining low flows and the corresponding influences of land cover changes.
- Effects of complex topography and variable vegetation (e.g., species, canopy structure) on precipitation reaching the ground, snowpack accumulation, and the snowmelt energy budget, and the corresponding influences on catchment-scale runoff processes.
- Effects of forest disturbance on runoff and groundwater response on the boreal plain, especially for sites underlain by fine-textured materials.
- Effects of forest disturbance on the occurrence and connectivity of concrete frost on boreal plain hillslopes.
- For many runoff generation processes that are generally well understood for specific research catchments, effort is needed to clarify the processes for a large range of catchment settings, which includes quantifying response thresholds and rates.
- Development of relatively simple models for planning and operational assessment purposes to supplement highly parameterized, physically based models that are focussed on research applications.

These knowledge gaps are manifested in the limited development of runoff indicators and associated thresholds. Application of some indicators is more of an art than a science (e.g., ECA thresholds) and most are a work in progress. These knowledge gaps result in an increased dependence on professional judgement for setting operational standards and evaluating disturbance impacts related to industry activities and watershed management. Without well-defined indicators and thresholds, professional judgement is at risk of elevated uncertainty and greater potential for error. As a result, a pressing need exists to refine hydrologic understanding, indicators, and thresholds for monitoring and managing cumulative effects on runoff.

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Climate Trend Mapper Available
Climate Trend Mapper computer program can be used to visualize climate data and trends recorded at weather stations across Canada. Sample output maps/graphs, and download Climate Trend Mapper at: http://climate.uwinnipeg.ca/

The Math You Need, When You Need It

Future Forest Ecosystems Scientific Council of British Columbia (FFESC)
The FFESC wrapped up its research program with a closing conference/workshop in June 2012. Draft synthesis and project summaries regarding the research are posted at: http://www.for.gov.bc.ca/hfp/future_forests/council/completed-projects

Mountain Pine Beetle Future Forests Webinar Series
Past webinars and presentations regarding post-beetle management and recovery efforts in the United States are available at: http://www.fs.fed.us/rmrs/events/future-forests/

Canadian Water Network International Conference on Assessing Pathogen Fate, Transport, and Risk in Natural and Engineered Water Treatment
September 23–26, 2012, Banff, Alberta
The assessment of pathogen fate, transport, and risk in natural and engineered water treatment is of great relevance to numerous applications, including riverbank filtration, aquifer storage and recovery, rapid sand filtration, water reuse, and evaluation of ground water under the direct influence of surface water. For more information, go to: http://www.cwn-rrc.ca/news-and-events/pathogen-fate-transport-risk-natural-and-engineered-water-treatment/

Keepers of the Water VI – Annual Watershed Gathering
September 26–29, 2012, Fort Nelson, B.C.
Fort Nelson First Nation will host this year’s Watershed Gathering.

For more information, go to: http://www.keepersofthewater.ca/gatherings/2012

Aquatic Toxicology Workshop
September 30–October 3, 2012, Kamloops, B.C.
Canada’s major annual meeting in the field of aquatic toxicity and related disciplines. It provides the opportunity to share information on current and emerging topics of regional, national, and international importance related to water quality. For more information, go to: http://www.atw.ca/

Resource Roads in British Columbia: Environmental Challenges at the Site Level
November 7–9, 2012, Cranbrook, B.C.
The environmental effects of roads are diverse, and include impacts on aquatic and terrestrial wildlife and habitat, soils, and water. At this conference, both road impacts and management responses will be addressed. For more information, go to: http://www.cmiae.org/Events/Resourceroads

12th Annual Stream Restoration Symposium
February 5–7, 2013, Stevenson, Washington
For more information, go to: http://www.rrnw.org/

BC Water and Waste Association Annual Conference & Trade Show
April 20–24, 2013, Kelowna, B.C.
For more information, go to: https://www.bcwwa.org/events/annual-conference.html

-- Print Version Available --

A print version is available for purchase through Crown Publications (http://www.crownpub.bc.ca). Digital versions of the two volumes or individual chapters can be downloaded from: http://www.for.gov.bc.ca/hfd/pubs/docs/Lmh/Lmh66.htm

You may also wish to follow the links at: http://www.forrex.org/publications/other

For further information, please contact: Robin Pike (Robin.G.Pike@gov.bc.ca) or Todd Redding (Todd.Redding@forrex.org).

**Recent Website Developments**

Plan2Adapt
Climate change impacts do not affect every region of British Columbia in the same way. The Plan2Adapt tool generates maps, plots, and data describing projected future climate conditions for regions throughout British Columbia. More information regarding this tool can be found at: http://pacificclimate.org/tools-and-data/pland2adapt

**Update**

Reminder

**Compendium of Forest Hydrology and Geomorphology in British Columbia**

http://www.forgov.bc.ca/hfd/pubs/Docs/Lmh/Lmh66.htm

You may also wish to follow the links at: http://www.forrex.org/publications/other

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**Recent FORREX Publications**


**Upcoming Events**

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3rd Annual Pacific Northwest Climate Science Conference
October 1–2, 2012, Boise, Idaho
The Pacific Northwest Climate Science Conference provides an annual forum to exchange scientific results and policy and management options related to climate change and climate impacts research. For more information, go to: http://pnwclimateconference.org/

Aquatic Toxicology Workshop
September 30–October 3, 2012, Kamloops, B.C.
Canada’s major annual meeting in the field of aquatic toxicity and related disciplines. It provides the opportunity to share information on current and emerging topics of regional, national, and international importance related to water quality. For more information, go to: http://www.atw.ca/

Water Supply Association of BC Annual General Meeting
October 18–19, 2012, Nelson, B.C.
For more information, go to: http://www.wsabc.ca/

Resource Roads in British Columbia: Environmental Challenges at the Site Level
November 7–9, 2012, Cranbrook, B.C.
The environmental effects of roads are diverse, and include impacts on aquatic and terrestrial wildlife and habitat, soils, and water. At this conference, both road impacts and management responses will be addressed. For more information, go to: http://www.cmiae.org/Events/Resourceroads

12th Annual Stream Restoration Symposium
February 5–7, 2013, Stevenson, Washington
For more information, go to: http://www.rrnw.org/

BC Water and Waste Association Annual Conference & Trade Show
April 20–24, 2013, Kelowna, B.C.
For more information, go to: https://www.bcwwa.org/events/annual-conference.html