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# A synthesis of the hydrological consequences of large-scale mountain pine beetle disturbance

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**Mountain Pine Beetle Working Paper 2010-01**

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## Abstract

The present mountain pine beetle (MPB) outbreak has become the most extensive ever recorded in the province of British Columbia. As of 2010, the cumulative area of provincial Crown forest affected was roughly 16.3 million ha. Due to the widespread abundance and commercial value of lodgepole pine, an aggressive program of salvage harvesting has been initiated, resulting in elevated cut rates in many of the areas affected by the outbreak. The MPB disturbance is located within the British Columbia Interior, an area characterized by a snow-dominated hydrologic regime. Within this regime forests play an important role in regulating the terrestrial water cycle, controlling how rain and snowfall are partitioned between interception, evaporation, snow and soil storage, runoff, and streamflow. Historically, most forest hydrology research into the effects of forest disturbance has been based on the impacts of clearcutting. The distinguishing feature of this MPB epidemic is that, despite its vast and unprecedented size, it is a biotic disturbance that does not necessarily result in complete stand loss. Unlike a stand-replacing disturbance (such as clearcutting or severe wildfire), even pure pine stands can retain a hydrologically functional secondary structure following beetle kill. The presence of such multi-storeyed secondary structure can mitigate the effects of beetle-kill. This issue compels better quantification of the impacts of a non-stand-replacing event on hydrology, and an improved understanding of how the hydrologic cycle is affected along a gradient of canopy loss and tree mortality.

The following synthesis is a review of research examining the effects of large-scale MPB-related disturbance conducted predominantly over the past five years. The emphasis is on research that explicitly examines the impact of beetle kill (or biotic disturbance in general) and the cumulative effects of large-scale salvage harvesting operations in response to beetle kill. In general, forest disturbance has been found to increase snow accumulation and melt, reduce interception loss and evaporation, and increase runoff and streamflow. There is a general consistency in the results, which show that biotic forest disturbance has a hydrologic impact intermediate between that of healthy forests and clearcut salvage harvesting. However, results are site specific and exceptions do occur, introducing variability and uncertainty in potential hydrologic impacts and making prediction difficult. At the basin scale, salvage harvesting is expected to increase water yield. Both beetle kill and salvage harvesting have the potential to affect the peak flow regime. When the impact is sufficiently large, the peak flow regime is adjusted such that for a given frequency of occurrence the magnitude is increased, and for a given magnitude the event becomes more frequent. Combined results suggest that beetle kill has a smaller impact on peak flow than the cumulative effect of both beetle kill and clearcut salvage harvesting. Research gaps remain; in particular, the ability to quantify the effects of secondary structure and stand deterioration and recovery over time is lacking.



**Keywords:** mountain pine beetle, salvage harvesting, hydrology, water balance, streamflow, hydrologic recovery

# Résumé

La présente infestation par le dendroctone du pin ponderosa (DPP) est devenue la plus étendue jamais répertoriée en Colombie-Britannique. En 2010, la superficie cumulative de forêts de la Couronne de la province infestées atteignait environ 16,3 millions d'hectares. Étant donné l'abondance et la valeur commerciale du pin tordu dans cette province, on a entrepris un programme intensif de coupes de récupération, ce qui a fait augmenter le taux de coupes dans de nombreuses zones touchées par l'infestation. Cette infestation sévit dans le Secteur intérieur de la Colombie-Britannique, une région caractérisée par un régime hydrologique dominé par la neige. Sous un tel régime, les forêts jouent un rôle important dans la régulation du cycle hydrologique terrestre; elles contrôlent la répartition de la pluie et de la neige entre l'interception, l'évaporation, le stockage sous forme de neige et dans le sol, le ruissellement et l'écoulement fluvial. Dans le passé, la plupart des travaux de recherche en hydrologie liés aux effets des perturbations forestières ont porté sur l'impact des coupes à blanc. La caractéristique marquante de la présente infestation par le DPP, malgré son étendue et son ampleur sans précédent, est qu'il s'agit d'une perturbation biotique qui n'entraîne pas nécessairement la perte complète des peuplements. Contrairement à une perturbation causant un rajeunissement de la forêt (comme les coupes à blanc et les incendies importants), même des peuplements constitués uniquement de pins peuvent conserver une structure secondaire fonctionnelle sur le plan hydrologique après la destruction partielle causée par le DPP. La présence d'une telle structure secondaire à plusieurs paliers peut atténuer les effets de la destruction des pins causée par le DPP. Cette question exige une mesure plus précise des impacts d'un événement non associés à un rajeunissement de la forêt sur l'hydrologie et une meilleure compréhension des effets allant de la perte de cime à la mort des arbres sur le cycle hydrologique.

L'étude de synthèse suivante passe en revue les travaux de recherche portant sur les effets d'une perturbation à grande échelle causée par le DPP, lesquels travaux ont surtout été effectués au cours des cinq dernières années. Nous mettons l'accent sur les recherches qui portent expressément sur l'impact de la destruction des pins causée par le DPP (ou la perturbation biotique en général) et sur les effet cumulatifs des opérations de récupération à grande échelle entreprises pour contrer cet impact. En général, les perturbations forestières augmentent l'accumulation de neige et la fonte, réduisent les pertes par interception et évaporation et augmentent le ruissellement et l'écoulement fluvial. Les résultats, qui sont en général concordants, indiquent que les perturbations forestières biotiques ont un impact hydrologique intermédiaire entre celui d'une forêt saine et celui d'une coupe de récupération. Toutefois, les résultats varient d'un endroit à l'autre et il y a des exceptions, ce qui introduit une certaine variabilité et incertitude dans les effets hydrologiques potentiels et rend les prévisions difficiles. À l'échelle du bassin, la coupe de récupération devrait augmenter l'apport d'eau. La destruction des pins par le DPP et les coupes de récupérations devraient avoir un effet sur le débit de pointe. Lorsque l'impact est suffisamment important, le débit de pointe varie de telle sorte que pour une fréquence donnée, l'ampleur augmente, et pour une ampleur donnée, l'événement devient plus fréquent. Ces résultats dans leur ensemble indiquent que la destruction des pins causée par le DPP a un impact moindre sur le débit de pointe que l'effet cumulatif conjoint de cette destruction et des coupes de récupération. Ces recherches comportent des lacunes; ainsi, il est difficile de mesurer les effets de la structure secondaire et ceux de la détérioration et de la récupération subséquente des peuplements avec le temps.



**Mots-clés :** dendroctone du pin ponderosa, coupe de récupération, hydrologie, équilibre hydrique, écoulement fluvial, récupération hydrologique

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# 1. Introduction

The mountain pine beetle (MPB; *Dendroctonus ponderosae* Hopkins) is a natural component of western North American forests. Although the MPB population normally exists at endemic levels, outbreaks do occur, and populations have reached epidemic levels several times over past decades within British Columbia (BC) (Taylor and Carroll 2004). Due to an abundance of mature (> 60 years) pine (Taylor and Carroll 2004) and a series of warmer than normal winters (Stahl et al. 2006), the present outbreak is of unprecedented severity and has become the most extensive ever recorded in the province (Wilson 2004). The BC Ministry of Forests and Range estimates that, as of 2010, the cumulative area of provincial Crown forest affected to some degree is about 16.3 million ha (BCMoFR 2010a). This infestation is expected to continue for at least another 10 years and kill at least 80% of the merchantable pine volume in the province (Westfall 2005). Trees that are successfully attacked are killed; this stops transpiration and leads to progressive defoliation and stand thinning within the affected area (Henigman et al. 2001; Lewis and Hartley 2006; Mitchell and Priesler 1998). Although the MPB attacks all pine species native to British Columbia forests, lodgepole pine (*Pinus contorta* Dougl.) is the most abundant species by area, and the most predominant commercial species. Consequently, an aggressive program of salvage harvesting has been initiated to recover as much economic potential as possible from dead timber and to mitigate the spread of MPB to high-risk areas (e.g., BCMoFR 2005). This has resulted in elevated cut rates in many of the administrative areas affected by the outbreak, and many salvage operations have created large (> 1000 ha) forest openings (Eng 2004).

The MPB disturbance is located within the BC Interior, an area characterized by a snow-dominated hydrologic regime (Eaton et al. 2002). Within this regime, forests play an important role in regulating the terrestrial water cycle: forests influence interception, evaporation, snow accumulation, snow melt, and ultimately soil storage and runoff (Adams et al. 1998; Bhatti et al. 2000; Hedstrom and Pomeroy 1998; Molotch et al. 2009; Pomeroy et al. 1998; Toews and Gluns 1986; Troendle and Reuss 1997). Historically, most forest hydrology research about the effects of forest disturbance has been based on the impacts of stand-replacing forest harvesting (i.e., clearcutting). The distinguishing feature of this MPB epidemic is that, despite its vast and unprecedented size, it is a biotic disturbance that does not necessarily result in complete stand loss. Unlike a stand-replacing disturbance (such as clearcutting or severe wildfire), a severe beetle infestation often leaves behind hydrologically functional secondary structure (juvenile pine, regeneration, understorey, non-pine canopy species, etc.; Coates et al. 2006), even in so-called pure pine stands. The presence of such multi-storeyed secondary structure can mitigate the effects of beetle kill (Schmid et al. 1991). This issue compels better quantification of the impacts of a non-stand-replacing event on hydrology and an improved understanding of how the hydrologic cycle is affected along a gradient of canopy loss and tree mortality. Is a beetle-killed forest a dead forest, or simply a living forest with dead trees and woody debris?

Historical studies on the effects of large-scale biotic disturbance are few, and those that do exist have mainly been empirical assessments of watershed-scale effects on streamflow. These studies generally indicate increased annual water yield, increased discharge during low flow periods, and increased spring runoff (Bethlahmy 1974, 1975; Love 1955; Mitchell and Love 1973; Potts 1984). There is no consistency between observed changes in peak flow: no detectable change was observed in two basins studied (White River, CO, Bethlahmy 1975; and Jack Creek, Montana, Potts 1984), but a 27% increase was observed in a third basin (Yampa River, CO; Bethlahmy 1975). Unfortunately, there are no watershed-scale studies that compare the incremental effects of large-scale insect disturbance to salvage harvest, so we don’t know whether these two disturbance types have the same hydrologic impact at the watershed and landscape scale. This issue remains a significant open question in light of the common management practice of aggressive salvage harvesting in the wake of large-scale biotic forest disturbance.

Sustainable forest management requires the ability to predict the consequences, and avoid or mitigate the negative consequences, of management decisions within the context of a dynamic forest landscape. One apparent consequence of the current beetle outbreak and associated salvage harvesting activities is alteration to the hydrologic cycle, which can manifest as changes in snow accumulation, soil moisture, and runoff at the local scale, and ultimately to changes in streamflow within larger watersheds. Changes to streamflow, both in magnitude and variation, can have attendant negative impacts on channel form and process, water quality, freshwater ecosystems, channel crossing or conveyance infrastructure, and public safety. Therefore, the purpose of this document is to synthesize research on the hydrologic impacts of MPB-related forest disturbance to inform sustainable forest management with respect to hydrologic effects.

A tremendous amount of literature has been devoted to investigating the impacts of forest harvesting over the past few decades, and several recent reviews for BC and western North America are provided by MacDonald and Stednick (2003), Moore and Wondzell (2005), Pike and Scherer (2003), Scherer (2001), Smerdon et al. (2009), and Winkler et al. (2010). This synthesis focuses strictly on work conducted after 2005 (i.e., since publication of Helié et al. 2005) that explicitly examines the impact of beetle kill (or biotic disturbance in general) or the cumulative effects of large-scale salvage harvesting operations in response to beetle kill. The recent research on the hydrologic effects of pine beetle and large-scale salvage harvesting is reviewed in Section 2. Stand-level studies reveal the physical processes governing the hydrologic effects of forest cover change and the subsequent impacts to the water balance at the local scale, and these are reviewed in Section 2.1. Although some management concerns at the local scale can make direct use of such information (e.g., availability of summer ground for forest operations), sustainable forest management typically involves understanding and predicting the cumulative effects of many stand-level disturbances spread across a complex landscape at a range of scales. This implies a necessitated understanding of the integrated effect of such changes to streamflow at some downstream location of interest. Unfortunately, translating stand-scale effects to the basin scale is not a trivial exercise due to the influence of the confounding factors of topography, morphology, channel network topology, climatic variability, spatial variability in forest cover, disturbance type, and intensity. A review of research examining basin-scale hydrologic effects is presented in Section 2.2, focusing on empirical (Section 2.2.1) and modelling (Section 2.2.2) approaches. This document concludes with a discussion of consequences and implications and suggestions for future research in Section 3.

# 2. Hydrological Response to Mountain Pine Beetle and Salvage Harvest

## 2.1 Stand-scale Hydrologic Effects

### 2.1.1 Snow Accumulation, Melt, and Ablation

Although many factors affect the spatial variability of snow accumulation and melt in the montane forests of British Columbia (such as climate, slope, aspect, and elevation), forest cover exerts a strong control (Jost et al. 2007; Packer 1962). Coniferous forest canopies can intercept a significant proportion of snowfall, which is subsequently lost to the atmosphere through sublimation, and this interception efficiency is strongly related to the density of canopy cover (Lundberg et al. 2004). Whereas snow accumulation differences between forest and open sites is predominantly the result of intercepted canopy snow lost to evaporation, differences in snow ablation are the result of differences in the snow surface energy budget. In this respect, forests act to absorb and reflect incoming shortwave radiation, emit longwave radiation, and dampen turbulent and latent heat fluxes (Adams et al. 1998).

Winkler et al. (2005) compared peak snow water equivalent (SWE) between three forest stands and a neighbouring clearcut (with 1-m-tall regenerating lodgepole pine). The three forest stands represented a range of forest structures: one site is a mature mixed species stand and the remaining two sites were juvenile (15 years old) lodgepole pine, with one site thinned to less than half the original stocking. Although none of the sites were reported to have beetle kill, the variation in stand density and canopy structure provides a proxy to the thinning effects of beetle kill. Differences in SWE between sites varied from year to year, likely as a function of the inter-annual variability of snowfall dynamics and mid-winter melt. Over 3 years of observation the ratio of forest-to-open SWE varied from 0.7–0.9 in the two juvenile stands to 0.6–0.7 in the mature, mixed-species stand (the SWE ratio for a clearcut is 1.0 by definition). The SWE difference between the clearcut and both mature and juvenile stands was statistically significant; however, there was no statistically significant difference in peak SWE between the two juvenile stands.

Beaudry (2007) compared snow accumulation in four stand types, collected at two sites, over several years near Prince George, BC: green mixed-species, green pine, grey-attack, and clearcut. Although relative differences were variable between sites, snow accumulation was generally highest in the clearcut, lowest in the green mixed-species stand, and intermediate in the grey-attack stands. In one year (2007) the grey stands had snow accumulation that was 17 and 49% greater than the neighbouring green mixed species stands, and the clearcuts had 45–87% more snow than the neighbouring green mixed species stands. Beaudry's results also suggest that snow accumulation in green pine stands is similar to that in a grey-attack stands, although only two green pine stands were available for sampling.

In a space-for-time study[1] Teti (2008, 2009) measured SWE in six groups of stands throughout the Interior Plateau of BC to directly quantify the impacts of beetle kill on snow accumulation and melt. Each group contained anywhere from four to six pine-leading forest stands, varying in age and canopy density, and a reference clearcut site. All the mature stands, and some of the juvenile stands, were affected by beetle kill with varying degrees of severity and both the red and grey stages of attack were represented. On a site-by-site basis, the SWE ratio was related to canopy density (indexed using the canopy gap fraction derived from hemispherical canopy photographs). The SWE ratio tended to decrease with increasing canopy density: a relationship that was statistically significant at four of the six sites. However, when pooling the data from all sites, no universal relationship between SWE ratio and canopy density was apparent, suggesting that snow accumulation dynamics show a strong geographic variability that overshadows canopy density effects (Teti 2009). For the mature beetle-attacked stands, canopy density generally decreased (and SWE increased) with increasing disturbance severity, whereas for the younger, healthy stands, canopy density tended to increase with increasing stand age (and SWE decreased; Teti 2008). On a site-by-site basis, SWE in the attacked stands was generally intermediate between those of the reference clearcut and nearby young, healthy stands.

In a smaller study near Vanderhoof, BC, Boon (2007) reported a similar result based on SWE measurements over a single year in live (healthy), dead (grey attack), and clearcut pine stands. Stand characteristics were as follows:

1. Healthy stand: 35-year-old 100% lodgepole pine with 80% canopy cover;
2. Grey stand: 100-year-old mixed species with 65% canopy cover; and
3. Clearcut stand: 10-year-old 100% lodgepole pine with less than 5% canopy cover.

[1] Many ecological processes occur on time scales that render experiment or observation impractical or difficult. A *space-for-time study* employs an experimental design whereby the observation of temporal dynamics over a long time scale (e.g., vegetation recovery and succession) is approximated using observations at multiple sites at various stages along a temporal gradient (e.g., multiple sites of varying forest age).

Peak SWE in the grey and healthy stands were 47 and 27% of peak SWE in the cleared stand respectively. Boon (2008) conducted a similar study over the 2006–2007 and 2007–2008 water years at a site near Fraser Lake in central BC. Again, peak SWE was compared between three plots: live, dead (grey attack), and clearcut. The difference in SWE between plots varied annually. In the high snowfall year of 2006–07, SWE in the forested plots (healthy and grey) was statistically larger than in the clearcut, but the two were not statistically different from each other. Snowfall during this year occurred during larger but less frequent events, and canopy interception differences between the healthy and grey stand were negligible. In the lower snowfall year of 2007–08, snowfall occurred over smaller but more frequent events, and canopy interception was more effective. In this case, average peak SWE differences between all stands were statistically significant, and SWE was highest in the clearcut and lowest in the healthy stand.

Winkler and Boon (2009) synthesized SWE and ablation[2] data collected from six different research projects (including those reported in Teti [2009] and Boon [2007, 2008]), quantifying SWE changes in lodgepole pine-dominated (> 50%) stands. The stands are located on the Interior Plateau of BC in the Sub-Boreal Spruce (SBS), Sub-Boreal Pine–Spruce (SBPS), and Montane Spruce (MS) biogeoclimatic zones within an elevation range of 730–1350 m above sea level (asl). Data was collected over the 2006–2008 period in stands experiencing a wide range of attack severity in the green-, red-, and grey-attack stages. These authors point out some of the inherent challenges and difficulties involved in these SWE–beetle disturbance research projects. Due to the unexpected speed of the infestation and the widespread attack of lodgepole pine, it has been necessary to establish retrospective experimental designs that lack the rigorous approach of the classic before–after/control–impact design (Stewart-Oaten et al. 1986). As well, most research plots have not been established for long enough to capture (or were established too late to capture) the transient nature of the disturbance (green- to red- to grey-attack), and a space-for-time approach was adopted, including using burned stands as surrogates for grey attack. Nevertheless, the pooled results are instructive and their summary is reproduced here and updated with an additional 2 years of data from Teti (2009). As per Winkler and Boon (2009), the data for SWE are categorized by stand type (old, mature, and intermediate) and attack phase (green/red and grey) and plotted in Figure 1. Because of the smaller number of sample points, the ablation data are pooled across all stand types. The aggregated results indicate that, on average, forested stands accumulate less snow than neighbouring openings (i.e., clearcuts), and green/red stands have less snow than grey stands (Figure 1). Forest/open ratios of average SWE for green/red stands are 0.72, 0.77, and 0.91 in the old, mature, and intermediate forest types respectively; for grey stands, the average SWE forest/open ratios are 1.0, 0.83, and 0.76 in the old, mature, and intermediate forest types respectively. There is considerable scatter in the individual results for all stand types and attack stages, and many ratios exceed a value of 1.0 (i.e., more snow accumulating under the forest than compared to the reference open site). Overall variability potentially derives from several sources, including: inter-annual variability in snowfall (i.e., differences in canopy snow interception between high and low snowfall years) (Harestad and Bunnell 1981) and mid-winter melt conditions (Gelfan et al. 2004), spatial variability in snowfall (differences between forest and clearcut, particularly if they are far apart) (Alila et al. 2009), differences in stand structure (e.g., canopy density, stand density, canopy height, stem spacing), and differences in survey methods (sampling rate and sample size per survey) (Winkler and Boon 2009).

There are several reasons that can explain observations of higher snow accumulation in a forest than a neighbouring clearing. These can include wind redistribution and increased sublimation within a clearcut

[2] *Ablation* is the process of snow removal, predominantly by means of melting, but also by such processes as wind scour and sublimation/evaporation. Experiments that measure snow removal by temporal sampling of snow water equivalent only are not able to distinguish snow melt from these other processes; hence, the decrease in snow water within such a sampling interval is referred to as *ablation*.

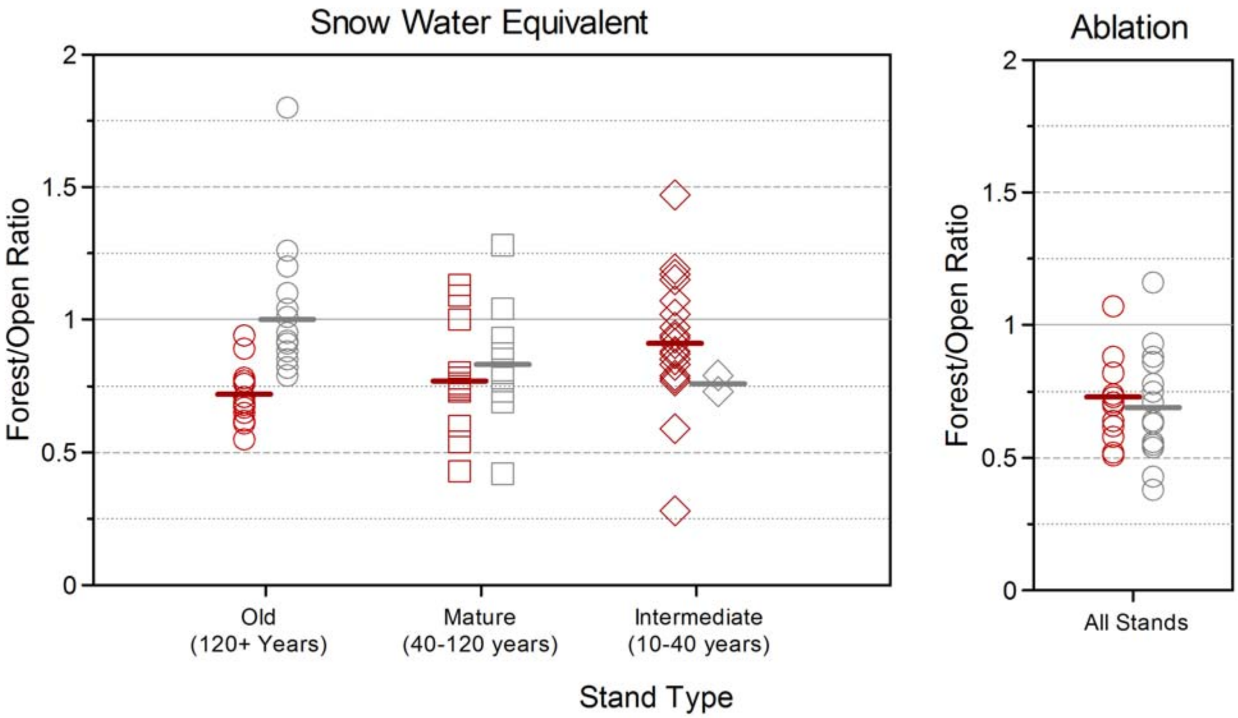


**Figure 1.** Summary of forest/open ratios for peak Snow Water Equivalent (SWE; left panel) and Average Ablation Rate (AAR; right panel) in pine-leading stands affected by the mountain pine beetle. Results for SWE and AAR are categorized by attack stage (red symbols indicate red/green attack; grey symbols indicate grey attack). The SWE data are categorized by stand age. Horizontal bars indicate sample averages. Adapted from Winkler and Boon (2009) with additional data added from Teti (2009).

due to high wind speeds and long wind fetch, particularly if the air is dry (Bernier and Swanson 1992; Alila et al. 2009). Large clearings that receive solar radiation throughout the accumulation season can also lose large amounts of snow water due to mid-winter ablation (Golding and Swanson 1986).

The data from Winkler and Boon (2009) and Teti (2009) indicate that green- and red-attack forest stands have lower average ablation rates (commonly estimated as ablation rate from peak SWE to snow disappearance) than neighbouring open areas (Figure 1). The average ablation rate forest/open ratio is 0.73 and 0.69 for green/red and grey stands respectively. Although the range in forest/open ratios is quite large (0.51–1.73 and 0.29–1.16 for green/red and grey stands respectively), most ratios are less than one; i.e., the average ablation rate is generally lower in the forest than in the open. Therefore, unlike the SWE forest/open ratios, the results for ablation more consistently suggest increased ablation rates in the open. Ablation ratios greater than 1.0 may derive from spatial variability between forested and open sites that are some distance apart (e.g., due to differences in snow accumulation, solar radiation, wind speed, and temperature). Wind redistribution of snow is not considered to be a factor during the spring snowmelt period as this phenomenon tends to be negligible for increasingly wet and cohesive snow (Alila et al. 2009).

Using meteorological and snow water equivalent measurements, Boon (2009) calculated the energy balance and simulated ablation rates in three 2500 $m^2$ study plots, healthy (green), grey, and clearcut, in a predominantly lodgepole pine forest stand near Fraser Lake, BC. Observations and simulations over the

2007 and 2008 melt seasons revealed that the average ablation rate is highest in the clearcut (10.9 and 13.7 mm/d), lowest in the healthy forest (8.0 and 8.5 mm/d), and intermediate in the grey stand (9.4 and 9.7 mm/d). Average forest/open ablation ratios were 0.68 and 0.79 in the healthy and grey stands respectively. Peak daily melt rates in 2007 and 2008 were highest in the clearcut (~28 and 30 mm/d) and lower, but similar in the healthy and grey stands (~14–15 mm/d in both stands) (estimated from Figure 5 in Boon [2009]). The date of peak melt in both years was the same in the healthy and grey stands, but occurred 10 and 6 days later in 2007 and 2008 respectively in the clearcut plot, indicating a change in melt (and likely runoff) synchronization between forested and open sites. In an earlier and similar study conducted near Vanderhoof, BC, Boon (2007) reported qualitatively similar results for the 2006 ablation season, where the average ablation rate was highest in the clearcut (4.9 mm/d), and similar in the forested stands (2.5 mm/d and 2.1 mm/d in the grey and green stands respectively). Based on energy balance modelling, Boon (2007, 2009) established that grey forest stands exhibit a snow pack energy balance different from both healthy and clearcut stands. The energy balance modelling results from Boon (2007, 2009) are summarized qualitatively (based on three snow melt seasons) in Table 1. A conceptual representation of the energy balance for a healthy forest stand during the melt period is show in Figure 2. Net shortwave radiation is positive (energy entering the snowpack) in all three stands. Due to loss of canopy cover, grey stands transmit more solar radiation than healthy stands, but not as much as a clearcut. Differences in canopy and stem density also affect net longwave radiation, which is negative (i.e., net energy leaving the snowpack) in the clearcut and grey stands, but negligible in the healthy stand. Net longwave radiation is an order of magnitude lower (more negative) in the clearcut than the grey stand. Both the healthy and grey stands have much lower below-canopy wind speeds than the clearcut, resulting in sensible and latent heat fluxes that are an order of magnitude lower than the clearcut, and of negligible magnitude. Net melt energy is dominated by shortwave radiation and sensible heat in the clearcut, but largely by net radiation in the forested plots. Net melt energy is highest in the clearcut, lowest in the healthy plot, and intermediate in the grey plot.

**Table 1**. Summary of the below-canopy energy balance for three stand types during the snowmelt period. The table shows the sign of the energy component for each stand type (shaded columns), and relative magnitude between stand types. Adapted from Boon (2007, 2009).

| Energy Component | Relative Magnitude between Stand Types | | | | |
|---|---|---|---|---|---|
| | Clearcut | | Grey | | Healthy |
| Net Shortwave | + | > | + | > | + |
| Net Longwave† | - | << | - | < | +/- |
| Sensible Heat Flux‡ | + | >> | + | ≈ | + |
| Latent Heat Flux† | - | << | - | ≈ | +/- |
| Net Energy | + | > | + | > | + |

† Values near zero in healthy stands; sign varies between years and sites.
‡ Values near zero in grey and healthy stands.

Teti (2008, 2009) demonstrated that ablation rate is directly related to the radiation transmittance (the proportion of above-canopy solar radiation transmitted through the forest canopy) of managed and infested lodgepole pine stands. Teti (2008) estimated transmittance based on analysis of hemispherical canopy photos. He shows that plot-average ablation rates increase linearly with plot-average radiation transmittance, and that radiation transmittance explained 64% of the variability in average forest/open ablation ratios across all study areas. These results show that, generally, ablation rates are highest in clearcuts (where radiation transmittance is 1.0 by definition), lower in recovering clearcuts, lower still in grey-, red-, and green-attack stands, and lowest in young, healthy stands. Teti (2009) also shows that a similar relationship can be observed between radiation transmittance and stand basal area (estimated as the square root of basal area of green- and red-attacked trees).

Although snow accumulation and melt are strongly influenced by forest cover, differences in snow accumulation and melt between forests and neighbouring openings are also affected by aspect and elevation (Packer 1962; Haupt 1979a, b; Toews and Gluns 1986; Jost et al. 2007). Therefore, in certain years or over large regions, forest cover may only be a second-order control compared to elevation (Alila et al. 2009; Jost et al. 2007). Based on snow water surveys between various stand types (clearcut,

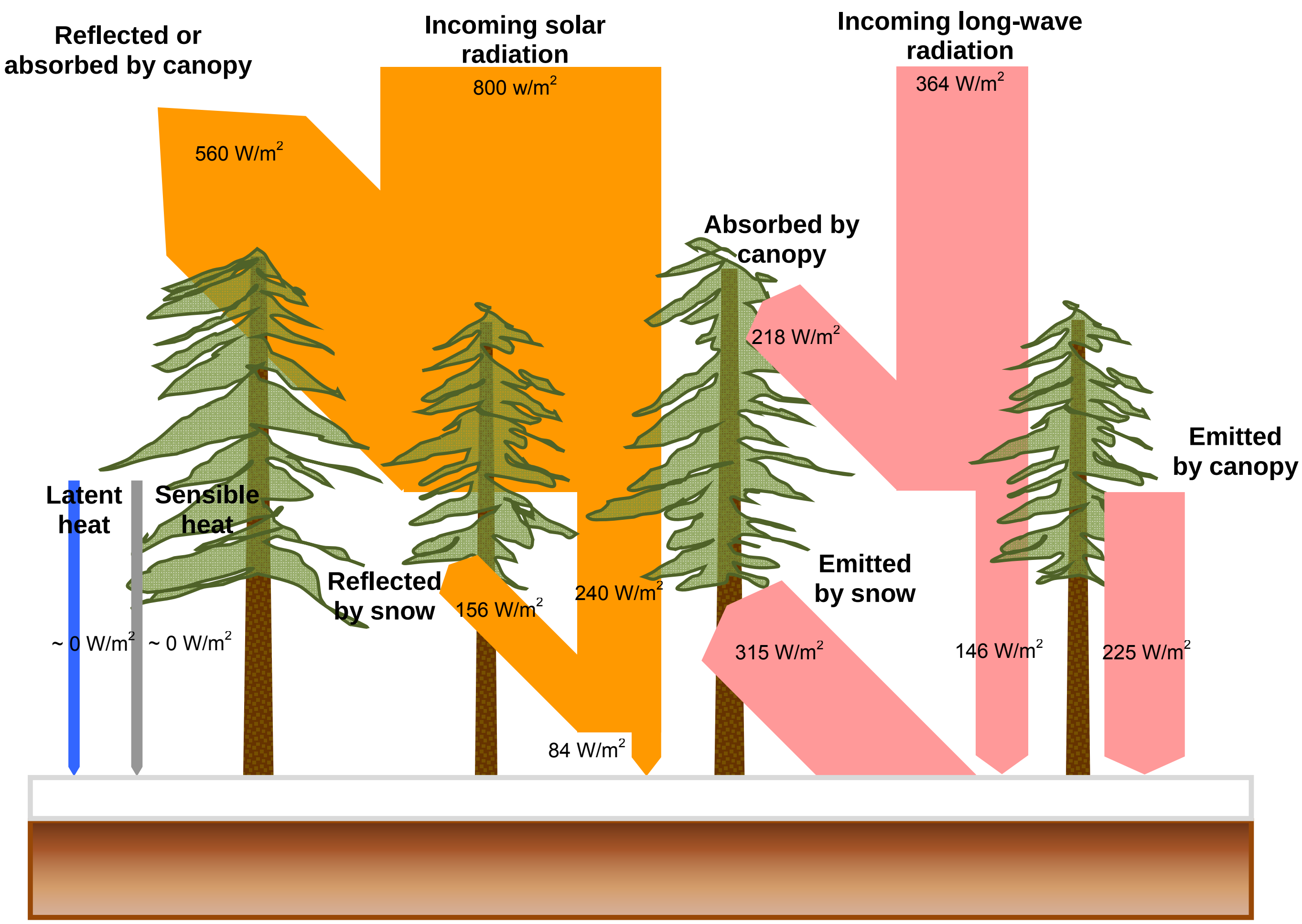


**Figure 2.** A conceptual graphical representation of the stand energy balance of a healthy mixed lodgepole pine-leading stand under clear sky conditions at midday during mid-April at a latitude of 50° north. Energy balance components and vegetation parameters are adapted from Boon (2009) and Teti (2008). Note that sensible and latent heat flux under a live forest canopy is typically negligible during the melt season due to very low wind speed. Ground heat flux (not shown) is also assumed to be negligible.

regenerating pine, red-attack, and grey-attack) in Baker Creek, BC, Alila et al. (2009) determined that elevation accounts for more spatial variation in SWE between stands than differences in forest cover; differences in SWE by elevation for a given stand type are larger than differences between stand types at a given elevation.

### 2.1.2 Water Balance

Spittlehouse (2006) used field observations and a process-based model to estimate stand water balance for stands of healthy lodgepole pine mixed with Engelmann spruce (*Picea engelmannii* Parry) and subalpine fire (*Abies lasiocarpa* [Hook.] Nutt). Modelling was based on data collected at Upper Penticton Creek (located at 1600–2100 m elevation in the Engelmann Spruce–Subalpine Fir [ESSF] biogeoclimatic zone, on the dry Okanagan Plateau in the Southern Interior of BC) from November 2002 to October 2005. Annual stand water balance from Spittlehouse (2006) averaged over the forested sites is shown conceptually in Figure 3 and summarized in Figure 4. During the study period, annual precipitation in the forested sites ranged from 584 mm in 2002–2003 to 774 mm in 2003–2004. Interception[3] (rain and snow) and total evaporation (transpiration from the canopy and understorey and soil evaporation) were fairly conservative from year to year, ranging from 21 to 23% of annual precipitation in 2003–04 and 32 to 35% in 2003–04. Consequently, interception and evaporation increased (or decreased) with increasing (or decreasing) annual precipitation, such that drainage[4] from the soil root zone varied from 211 mm (36% of annual precipitation) in a dry year to 360 mm (46% of annual precipitation) in a wet year. Total evaporation was broken down into 71% tree transpiration, 14% understorey transpiration, and 15% soil evaporation. The residual change in soil moisture storage was quite small, ranging from 37 to -16 mm (6 to -2% of annual precipitation).

Silins et al. (2007) provide a synthesis of stand water balance observations and modelling of healthy lodgepole pine-dominated stands characteristic of north and west-central Alberta. This region is characterised by an average of 598 mm of precipitation over the 10-year study period (1985–1994), with a range of 538–749 mm. Water balance results are based on the rainfall-dominated portion of the season (April 15 to October 14), which forms the majority of annual precipitation in north and west-central Alberta. Based on simple water balance modelling, Silins et al. (2007) report that canopy transpiration is, on average, 30% of annual precipitation (ranging from 23 to 38%). Total daily canopy transpiration averaged between 1.5 and 2 mm/day with midday peaks equivalent to 4–5 mm/day. Respectively, average canopy and litter rainfall interception were 18% (ranging from 13 to 21%) and 26% (ranging from 22 to 29%) of total annual precipitation. Forest floor rainfall interception was higher than canopy rainfall interception. Average drainage was reported as 29% of annual precipitation, ranging from 21 to 36% over 10 years. In absolute terms, canopy transpiration is the most conservative quantity, varying between 159 and 188 mm over 10 years (with an average of 174 mm), whereas drainage was sensitive to annual precipitation and ranged from 116 to 255 mm over 10 years (with an average of 174 mm). Canopy interception averaged 105 mm, ranging from 87 to 122 mm, and litter interception averaged 152 mm, ranging from 131 to 185 mm.

Based upon observations in several different lodgepole pine-dominated stand types in the BC Interior, Carlyle-Moses (2007) determined that rainfall interception is a function of stand structure. In a mature forest stand (a mix of pine, spruce, and fir), total growing-season interception increases with increasing canopy cover fraction, whereas in a juvenile stand (pine dominated), total interception increases with

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[3] *Interception* in this context refers to precipitation (rain and snow) intercepted by and stored within the forest canopy, and subsequently lost to the atmosphere by evaporation.

[4] *Drainage* is water that exits the stand by vertical percolation, regenerating groundwater and/or becoming streamflow, and typically moving to the nearest stream channel as sub-surface flow.

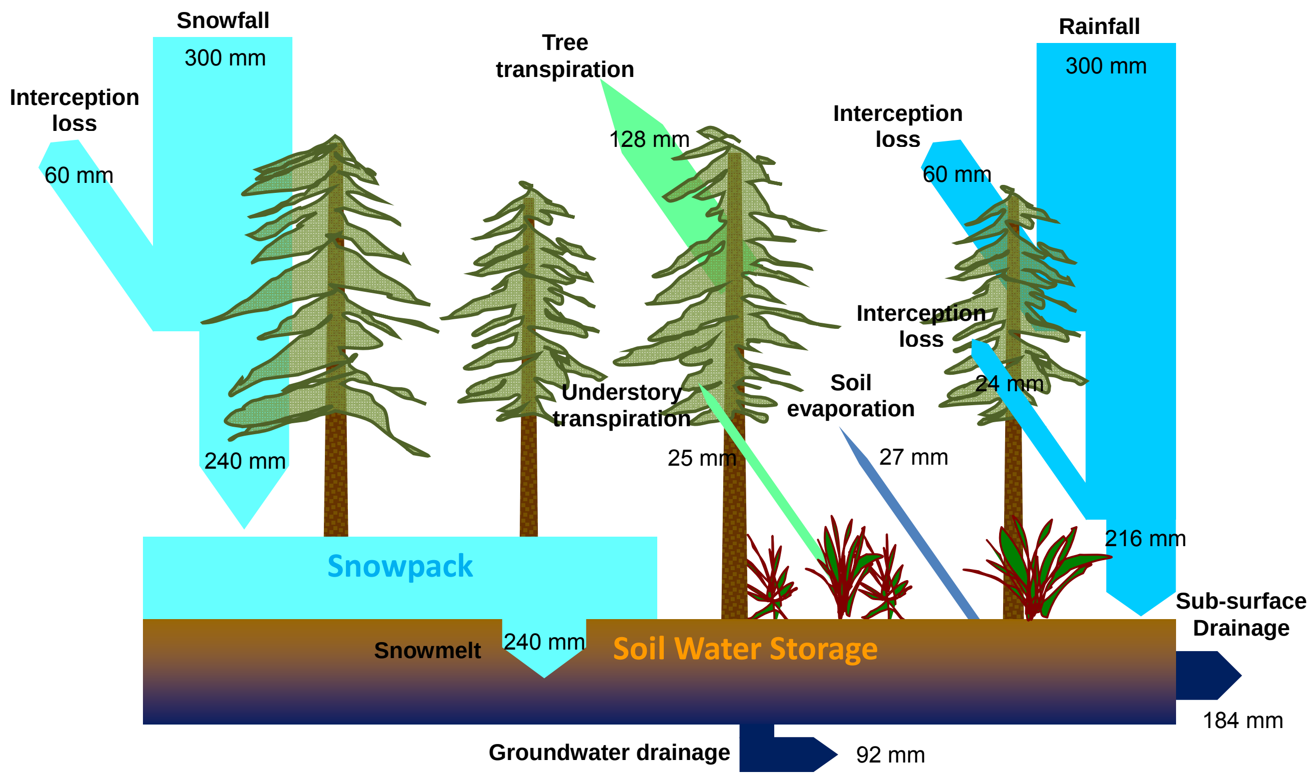


**Figure 3**. A graphical representation of the average annual stand water balance for a healthy, mature, mixed pine-leading stand. Relative magnitudes are adapted from Spittlehouse (2006), based on a process-based water balance modelling of a mixed lodgepole pine–Engelmann spruce–subalpine fir stand at Upper Penticton Creek, BC. Winter snow ablation is assumed to be negligible. Soil drainage is assumed to partition into two-thirds sub-surface runoff and one-third groundwater drainage. Understorey interception is assumed to be 10% of throughfall and active only in the snow-free period.

increases in both canopy cover fraction and basal area. Total rainfall interception (measured from June 18 to August 4) was 33% and 41% of rainfall in the mature and juvenile stands respectively. Carlyle-Moses (2007) also points out the importance of forest floor interception in growing-season stand water balance, showing that interception by the bryophyte moss layer may be equal in magnitude to forest canopy interception.

Following beetle disturbance or salvage harvesting, changes in canopy structure and reduction of canopy transpiration affects various components of the stand water balance. These various changes are qualitatively summarized for healthy, red-attack, grey-attack, and clearcut stands in Figure 5. The results of Figure 5 are predominantly synthesized from the stand water balance modelling of Spittlehouse (2006, 2007), with stand types based on those used by Spittlehouse (2007). The prototype healthy forest is modelled on a mature lodgepole pine stand at Upper Penticton Creek (100+ years of age, 50% canopy cover, and 20–22 m tall; the understorey is a mixture of 40% small shrubs, forbs, and grass and 60% mosses and lichens, litter, and bare soil). The red-attack stand assumes all forest trees in the healthy stand are dead (100% attack), but needles and understorey are retained. The grey-attack stand assumes all the needles are lost, and the clearcut is presumed to be devoid of all vegetation and ground cover. As already discussed, canopy snow interception tends to decrease with increasing disturbance severity, such that snow accumulation tends to decrease along the gradient of clearcut to grey to forested (red and healthy) stands (Beaudry 2007; Boon 2007, 2008; Spittlehouse 2007; Teti 2009; Winkler and Boon 2009).

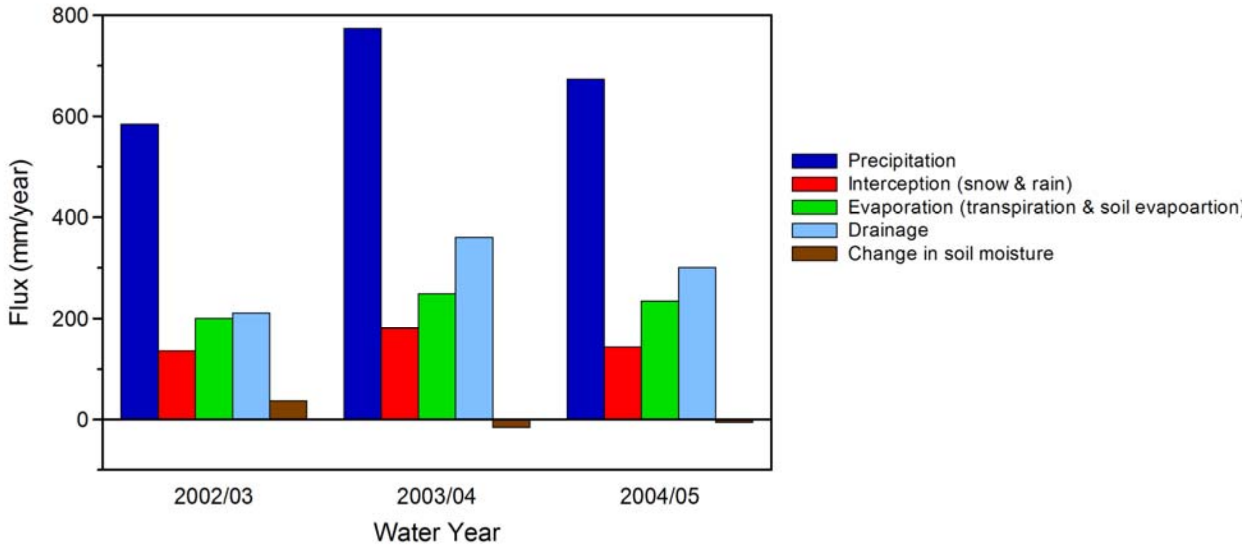


**Figure 4**. Annual average stand water balance for a healthy mixed forest stand of lodgepole pine, Engelmann spruce, and subalpine fir. Data from Spittlehouse (2006).

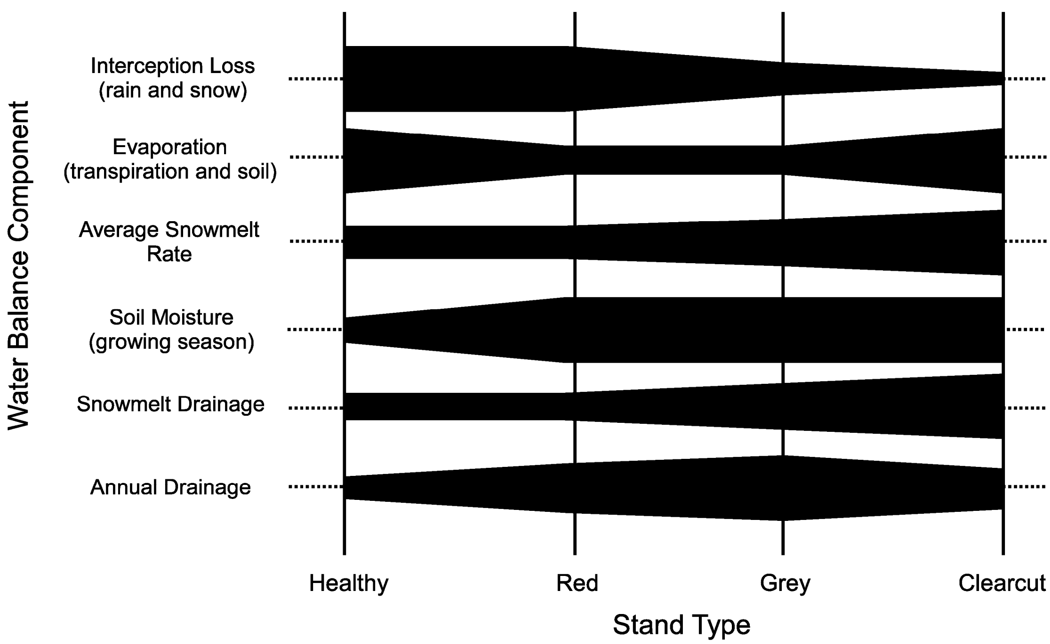


**Figure 5**. Qualitative summary of stand water balance along a disturbance gradient represented using four generic forest stand types. The width of each polygon figure shows the relative proportion of water balance component for each stand type. Note that figure widths are not to scale between water balance components. Adapted from Beaudry (2007), Boon (2007, 2009), Carlyle-Moses (2007), Spittlehouse (2006, 2007), Teti (2009), and Winkler and Boon (2009).

Rainfall interception is highest in healthy and red-attack forest stands, which have similar cumulative interception properties. Although no direct observations of rainfall interception in grey stands could be found, Carlyle-Moses (2007) observed that a burned forest stand, which is cautiously used as a proxy for a grey-attack stand, has rainfall interception of only 8% of growing season rainfall, as opposed to the 29% estimated for a nearby mature lodgepole pine stand. Interception in grey stands has also been modelled at levels as high as 50% of forest interception (Spittlehouse 2007). Spittlehouse (2007) observed that, following the snowmelt period, the soil water content in clearcuts was greater than that in adjacent forested sites, with the greatest difference occurring during the late summer and early fall. Soil remains wet in the clearcut due to negligible interception loss, loss of both understorey and tree transpiration, and replenishment from summer and fall rain. However, wetter soil in the clearcuts resulted in annual soil evaporation of similar (if not higher) magnitude to total evaporation (canopy transpiration, understorey transpiration, and soil evaporation) in the forested sites. Soil also remains wet in the red- and grey-attack stands due to loss of tree transpiration and to limited transpiration from the partially-shaded understorey (Spittlehouse 2007). Differences in drainage are due to differences in interception and evaporation between stand types (Figure 6). Consequently, drainage during the snowmelt period tends to increase following stand disturbance: the clearcut has the highest drainage during the snowmelt period, the lower interception capacity of the grey-attack stand results in more drainage than the forest and red-attack stands, and the red-attack stand has slightly higher drainage than the healthy forest (Spittlehouse 2007). The increase in spring snowmelt drainage compared to the healthy forest depends on the antecedent soil moisture status in the forest from the previous summer and fall period. Drier forest soil (due to low amounts of summer and fall rain) reduces and delays drainage from the forest, causing a larger relative increase between the forest and perennially wet soil of the red, grey, and clearcut stands (Spittlehouse 2007). On an annual basis drainage is highest in the grey stand, lowest in the healthy forest, and intermediate in the red and clearcut stands (Figure 6). Spittlehouse (2007) also conducted simulations for various percentages of tree mortality: he noted that the red-attacked stand, which had greater than 60% live trees interspersed with dead trees, had similar drainage and summer root zone soil water content to that of the healthy forest.

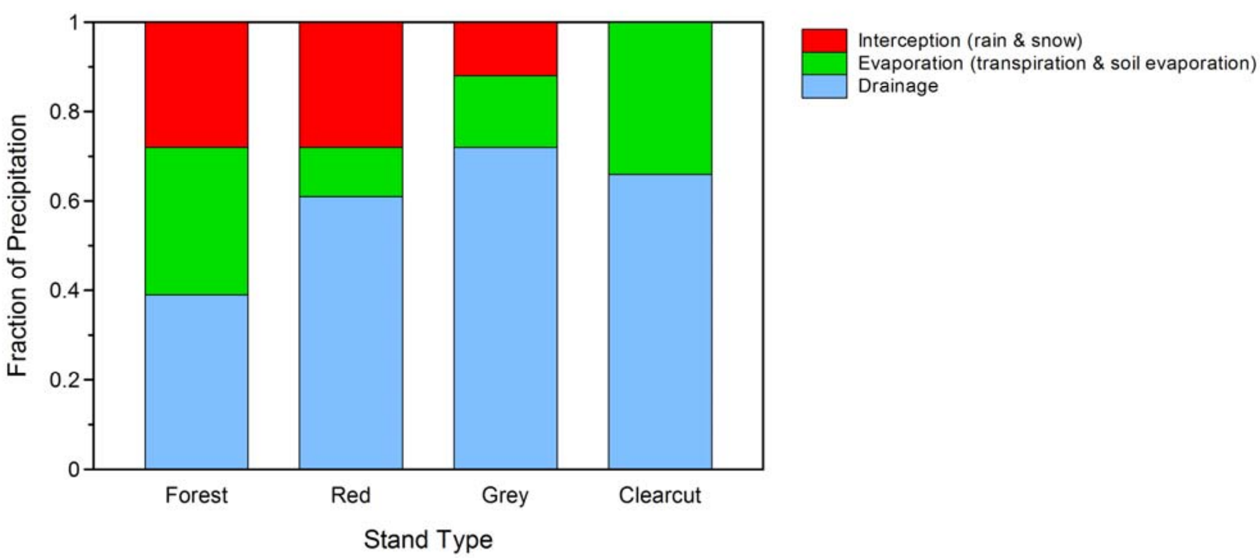


**Figure 6**. Mean annual stand water balance for various lodgepole pine stand types, based on results from a stand water balance model run over the period from October 2002 through September 2006 with meteorological and stand data collected at Upper Penticton Creek. The model was calibrated to the healthy and clearcut stands; parameters were estimated for the red and grey stands. Adapted from Spittlehouse (2007).

### 2.1.3 Transient Forest Cover and Hydrologic Recovery

Hydrologic recovery is the term used to describe the return of a disturbed forest stand over time to its pre-disturbance hydrologic function (i.e., rain and snow interception, snow melt, transpiration). In the short-term, the two extremes of the recovery spectrum are the pre-disturbance healthy forest and the stand-replacing clearcut. Unlike a stand-replacing disturbance, even so-called pure pine stands, and certainly mixed-species stands, can retain substantial amounts of residual secondary structure following a beetle-kill event (e.g., juvenile pine, regeneration, understorey, other canopy species; Axelson et al. 2009; Coates et al. 2006, 2009; Nigh et al. 2008; Vyse et al. 2009). In such cases, disturbed stands may already, relative to stand-replacing events, retain some hydrologic function (Schmid et al. 1991) and be "partially recovered." Nevertheless, all forest disturbance effects are transient in nature, such that the hydrologic impacts vary through time as stands continue to deteriorate and/or regenerate following disturbance.

The rate of hydrologic recovery in a beetle-killed stand will be determined by the relative and off-setting rates of canopy degeneration (defoliation and fall down of dead trees) and canopy regeneration from secondary structure (Boon 2009; Lewis and Huggard 2010). For example, Silins et al. (2007) observed that partial canopy removal in lodgepole pine stands reduces canopy transpiration in the short term, but after 5 years crown growth and physiological adjustments in the remaining trees result in recovery of nearly half of the transpiration that was initially lost. This finding suggests that in beetle-killed stands, transpiration recovery can potentially occur over a short period of time if sufficient secondary structure exists in the stand. In his space-for-time study, Teti (2009) makes inferences about relative snow accumulation and ablation recovery rates of managed (recovering clearcuts) and beetle-killed stands. His data, although preliminary, indicate that managed stands experience decreasing snow accumulation and ablation rates through time, due to increasing canopy density and decreasing radiation transmittance. Beetle-killed stands, on the other hand, appear to experience increasing snow accumulation (but decreasing ablation rates) through time due to changes in stand structure brought on by defoliation and degeneration of the dead canopy (decreasing canopy density and increasing transmittance), possibly offset by the presence of secondary structure (Teti 2008). Teti (2009) infers that the "crossing point" between increasing canopy density in managed stands and decreasing canopy density in killed stands occurs at about 20–25 years after disturbance (i.e., it takes a managed clearcut 20–25 years before the hydrologic impact is less than that of an unsalvaged stand). Consequently, although harvesting has a larger initial effect on snow accumulation and ablation than beetle kill, the effects of harvesting on snow hydrology could become negligible sooner than those effects in beetle-killed stands.

Huggard (2008) and Lewis and Huggard (2010) describe a model to estimate recovery rates of clearcut and beetle-killed stands. They model stand recovery in terms of equivalent clearcut area (ECA), which is a common indicator used to assess potential hydrologic impacts of forest cover loss relative to a clearcut (100% ECA) and a mature stand (0% ECA). ECA is commonly used to assess management impacts on spring runoff and peak flows and, as such, is mainly a measure of snowpack recovery (BCMoF 2001). The model explicitly incorporates: a) the deterioration in hydrologic function of dead trees over time as needles and branches are lost and dead snags fall; b) the contribution of remaining live non-pine overstorey and understorey trees; and c) recovery due to the expected rate of growth and mortality of surviving non-pine canopy trees, saplings, and seedlings along with natural ingress (Lewis and Huggard 2010). They compared the recovery rates of beetle-killed stands to those of managed clearcut and partially harvested stands (i.e., salvage logging plus planting) in a range of biogeoclimatic sub-zones and variants. Their results indicate that in a range of pine-leading stands the immediate (up to one year following disturbance) ECA of salvage logging (100% ECA) is always higher than that in unsalvaged, beetle-killed stands. For the range of stand types examined, the initial ECA of an unsalvaged stand is generally between 10 and 20%. However, the ECA of unsalvaged stands increases over time, dominated by the degeneration of the dead pine trees, peaking between 10 and 20 years following disturbance, after which ECA declines with increasing growth of understorey saplings and seedlings. Full recovery (0% ECA) of unsalvaged stands is estimated to take anywhere from 40 to 60 years, and peak ECA ranges from 40 to over 70% (Huggard 2008). Figure 7 shows an example depiction of ECA evolution through time for a pine-leading stand in the Montane–Spruce (MS) biogeoclimatic zone with an understorey of 50% spruce and 50% subalpine fir, adapted from Huggard (2008). The evolution of ECA in unsalvaged stands is sensitive to many factors, including the presence and amount of non-pine canopy trees, presence and amount of understorey trees, release and regeneration delay, and the growth and survival rate of understorey trees and seedlings (Lewis and Huggard 2010). Recovery of planted clearcuts can be quite rapid, declining from 100% ECA at year 0 to full recovery within 20–35 years. The ECA following salvage logging plus planting can decrease below that for an unsalvaged stand in the same forest type at any time between 10 and 30 years following disturbance, which is generally consistent with the timeframe suggested by Teti (2009) based on canopy density observations. The recovery rate of a partially salvaged forest stand is estimated to be intermediate between that of an unsalvaged stand and full-salvage-logged stand (e.g., Figure 7).

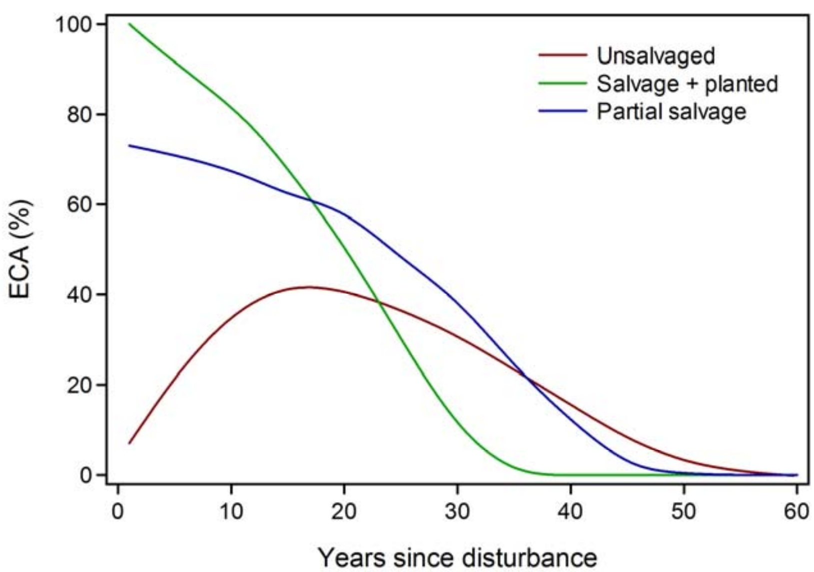


**Figure 7**. Conceptual depiction of ECA evolution through time for unsalvaged, salvaged plus planted, and partially salvaged plus planted pine-leading stands in the MSdm3 sub-zone with an understorey of 50% spruce and 50% subalpine fir. Adapted from Huggard (2008).

The recovery rates given by Huggard (2008) and Lewis and Huggard (2010) are conceptual. As stated earlier, ECA is only an index of snow process recovery (which pertains to peak flow recovery), and it has not been found to correlate well with observed peak flow impacts at the watershed scale (Scherer 2001). In the context of Figure 7, ECA is, more directly, a description of changes in composite stand height and canopy density with time. Also, other components of the stand water balance (e.g., soil evaporation, tree transpiration, soil moisture, and annual drainage) may not follow the same recovery trajectories as those estimated for snow accumulation and melt. Nevertheless, Lewis and Huggard (2010) effectively show that the effects of disturbance are transient, and the labels of red and grey attack are overly simple and don’t do justice to the wide range of stand types and complexity that exist during and following a beetle attack.

## 2.2 Basin-scale Hydrologic Effects and Streamflow Changes

### 2.2.1 Empirical Studies

Over the past 5 years there have been no empirical watershed-scale studies conducted in BC on the explicit effects of beetle kill on streamflow, although two studies have investigated the impacts of large-scale salvage harvesting. In response to a spruce bark beetle infestation in 1975, the Bowron River watershed (3590 $km^2$) was subject to a cumulative harvest of approximately 900 $km^2$ (25% of the basin area) over a 10-year period. The neighbouring Willow River watershed (3110 $km^2$) was also subject to large-scale forest harvesting over a cumulative area of 1064 $km^2$ (34% of the basin area), but during a longer 51-year period (1954 to 2005). Wei and Lin (2007) and Lin and Wei (2008) examined the impact of this large-scale forest harvesting on seasonal, annual, low, and peak flows in both watersheds by comparing the streamflow and ECA time series from 1954 to present in each watershed. Their results were mixed and varied by watershed. Results were generally inconclusive for the Bowron watershed, with no significant changes in the annual peak flows, spring freshet, mean annual flow, mean summer flow, or mean winter flow (Wei and Lin 2007). Results indicate the Willow watershed is more sensitive to forest harvesting, and showed significant increases in the annual maximum peak flows, spring freshet (April through June) runoff, and mean annual flow (Lin and Wei 2008). Lin and Wei (2008) specifically controlled for inter-annual peak flow variability due to climatic effects (by using snowfall and air temperature). Nevertheless, mean summer (July through October), winter (November through March), and annual low flows in the Willow watershed appear to be unaffected by forest harvesting (Lin and Wei 2008). However, the validity of using ECA as a quantitative measure of hydrologic recovery at the watershed scale remains questionable, since ECA alone doesn't explain why the two watersheds respond so differently to forest harvesting, nor why the Bowron watershed is apparently so insensitive to intensive salvage harvesting.

Stednick and Jensen (2007) studied the impact of a pine beetle epidemic on a number of watersheds in Colorado and Wyoming. They compared watersheds containing a wide range in the amount of disturbed area to nearby control watersheds that were relatively undisturbed using analysis of covariance. Preliminary observations indicate that the response of annual water yield following beetle kill is highly variable, and has no consistent relationship with the percent of area disturbed. Annual water yield changes ranged from more than 30% higher water yield to reductions of as much as 20% following beetle kill. They observed that water yield increases were associated with beetle kill in even-aged forests that typically lack any significant understorey vegetation. Conversely, where watersheds are composed of mixed forest (mature lodgepole pine with other species in the understorey), they hypothesized that the remaining non-pine vegetation responded to increased soil moisture with increased growth rate, resulting in reduced water yields. The preliminary results for watersheds studied by Stednick and Jensen (2007) showed a mixed response with respect to changes in annual maximum peak flow. In some cases peak flows were increased, and in other cases no increases were detected; regardless, none of the changes were considered statistically significant.

### 2.2.2 Modelling Studies

Two early modelling studies in the post-2005 period focused exclusively on the effects of salvage harvesting. Alila and Luo (2007) used the UBC Watershed Model (UBCWM) to simulate the effect of clearcut harvesting in nine watersheds; eight located in the Okanagan basin and one on the Fraser Plateau in BC. The experiment compared long-term (22–76 years, depending on the available climate data) simulated streamflow of a fully forested baseline scenario to a scenario of 100% clearcut harvesting. While 100% salvage harvesting is admittedly unrealistic in many large watersheds (although it is potentially realistic for watersheds of 10 $km^2$ or less; Eng 2004), it is useful to provide an upper limit on possible hydrologic impacts. Based on their model results, Alila and Luo (2007) indicate that 100% clearcut harvesting results in statistically significant (95% confidence level) increases in annual maximum peak flow across a wide range of return periods[5]. Across the watersheds studied, peak flow increases ranged from 30% to 100% for 1- and 2-year events and from 35% to 75% for 10- and 20-year events. Changes in freshet (May, June, and July) runoff ranged from 40% to 75%, where 60% of the increased water yield was attributed to the absence of forest canopy interception loss and the remaining 40% was attributed to an absence of tree transpiration. The watersheds studied all have the same biogeoclimatic characteristics, differing only in local physical attributes (Alila and Luo 2007). Consequently, the impact of forest harvesting on streamflow could be directly related to the physiographic characteristics of the basin. Using multivariate regression, they determined that 86% of the between-watershed variation in relative change in peak flow and water yield could be explained by basin size (negative correlation), mean basin slope (positive correlation), and equivalent forested area (i.e., relative forest area multiplied by watershed-average crown closure) (positive correlation).

Rothwell and Swanson (2007) report on the results of using the Water Resources Evaluation of Non-point Silvicultural Sources (WRENSS) model to predict the impact of salvage harvesting on streamflow originating from several sub-basins of the Smoky, Simonette, Muskeg, and Little Smoky Rivers (sub-watershed areas vary from 20 to 249 $km^2$). This region near the town of Grande Cache, Alberta receives annual precipitation of 590 mm and generally experiences annual water yield of 225–250 mm. Rothwell and Swanson (2007) provide initial results for a scenario that reflects a salvage harvest rate of 75% of mature pine stands over 20 years. Annual water yield is projected to increase from 4 to 30% between years and between watersheds, and maximum water yield increases occurred in watersheds where over 50% of the watershed area was harvested. Low to moderate increases occurred when 8 to 43% of the watershed area was harvested.

The Forest Practices Board (2007) and Alila et al. (2009) used the Distributed-Hydrology-Soil Vegetation Model (DHSVM) to explicitly assess the effect of beetle kill, and the combined effect of beetle kill and salvage harvesting, in Baker Creek. Baker Creek is a 1570 $km^2$ basin located in the Central Interior of BC (the creek is a tributary of the Fraser River) that is almost entirely forested, predominantly by lodgepole pine (85% of the forest area). As of 2007, 75% of the mature lodgepole pine in the basin was infested by MPB (Forest Practices Board 2007). A number of scenarios were investigated, ranging from baseline forest cover (circa 1970; minimal harvesting and no beetle kill) to pre-beetle conditions (1995; 15% harvest above baseline and no significant MPB) and the effects of beetle kill (2005; 21% harvest over baseline and 80% of pine-leading stands infested), plus a range of dead pine salvage harvesting options (circa 2005 and later, with 60–0% forest retention, where retention includes all live spruce plus dead pine). In general, Alila et al. (2009) report that annual maximum peak flow was projected to increase with increasing cumulative disturbance and that, for a given disturbance scenario, the relative increase in peak flow from baseline conditions was constant across all return periods (from 2- to 50-year events). Peak flow increases ranged from 66 to 104% for the 2-year event, 65 to 99% for the 10-year event, 65 to 99% for the 20-year event, and 77 to 117% for the 50-year event (all changes were statistically significant at the 95% confidence level). The exception is the pre-beetle scenario which, although it indicated increases in peak flow due to conventional logging from 1970 to 1995, did not project statistically significant changes (Forest Practices Board 2007). Alila et al. (2009) attribute the dramatic peak flow changes to both the relatively large area of the watershed affected by forest disturbance and to the low relief of the basin, which generally leads to basin-wide synchronized melt and runoff reaching the stream network. Freshet water yield (April through July) was projected to increase with increasing cumulative disturbance,

[5] The *return period* of an event (e.g., annual maximum peak flow) is often used in place of stating the exceedance probability. The annual maximum peak flow exceeded with a 1% probability in any year, or a chance of 1 in 100, is called the 100-year event; the annual maximum peak flow exceeded with a 2% probability in any year, or a chance of 1 in 50, is called the 50-year event, and so on.

ranging from 7% for the pre-MPB scenario to 55% for worst case disturbance scenario (100% harvest of the entire basin). All changes in the freshet water yield (with the exception of the pre-MPB scenario) are significant at the 95% confidence level.

As of 2007 approximately 8 million ha of forest within the Fraser River basin, or 35% of the drainage area, was affected by pine beetle infestation. This situation motivated a study that used a hydrologic model to assess the impacts of MPB-related forest disturbance within the Fraser River watershed (Schnorbus et al. 2010). Using the Variable Infiltration Capacity Model (VIC), Schnorbus et al. (2010) assessed possible changes in annual maximum peak flow in 60 sub-basins of the Fraser River, ranging in area from 330 to 217 000 $km^2$. The cumulative impact of beetle kill and clearcut salvage harvesting was assessed using seven disturbance scenarios: a pre-infestation baseline forest cover (circa 1995); current forest cover (circa 2007); and five hypothetical scenarios of increasing disturbance severity beginning with the baseline scenario plus 100% kill of all mature pine , and then progressing through salvage harvest rates of 25, 50, 75, and 100% (by area) of dead pine. Findings from Schnorbus et al. (2010) were qualitatively similar to those of Alila et al. (2009) in that forest disturbance tends to cause the peak flow frequency curve to shift upward such that for a given return period, the magnitude of the event is increased, and for a given magnitude, the return period decreases (the event occurs more frequently; Figure 8). Changes in recurrence interval can often be larger than changes in magnitude. This can be particularly evident in snowmelt-dominated watersheds where the slope of the frequency curve is quite low. For example, in Baker Creek the magnitude of the 20-year peak flow event is predicted to increase by approximately 25% (for a scenario of 100% pine mortality plus 25% salvage harvesting of dead pine), but the recurrence interval is predicted to decrease from 20 years to 7 years (i.e., more than double in frequency; Figure 8a). The relative change in quantile magnitude increases with increasing disturbance severity. The cumulative effect of forest harvesting and beetle kill manifests a stronger effect than beetle kill alone; the impact is generally lowest for current conditions (i.e., 2007) or the hypothetical 100% beetle kill.

Schnorbus et al. (2010) also show that the sensitivity of the peak flow regime to forest disturbance exhibits large spatial variability, as demonstrated using the 1-in-20 year quantile in Figure 9. Relative changes (from baseline) in the magnitude of the 20-year peak flow event range over the studied basins from no change to a) 8% for current forest cover, b) 8% for 100% beetle kill (0% harvesting), c) 46% for 25% harvesting (Figure 9), d) 91% for 50% harvesting, e) 130% for 75% harvesting, and f) 172% for 100% harvesting of dead pine. Schnorbus et al. (2010) suggest that this spatial variability is related to both the relative size of the disturbance area and the proportional runoff from any pine-covered areas within each respective sub-basin. The increase in peak discharge was shown to increase with increasing disturbance area; thus, sub-basins with large proportions of their area covered in mature pine are (subject to actual infestation and harvest rates) more vulnerable to impact than basins with little or no pine. However, if proportionately little runoff for a sub-basin derives from the pine-covered area, then such a sub-basin can be relatively robust to beetle-related forest disturbance. Generally, small sub-basins with subdued topography (i.e., with no significant alpine/sub-alpine snow contribution) and with large proportions of susceptible pine (and, therefore, large disturbance areas), such as those found in the semi-arid Fraser Plateau (i.e., Baker Creek, Figure 8a), appear to be the most sensitive to forest disturbance. Sub-basins without significant pine-forested areas (and, therefore, small disturbance areas) and/or sub-basins which integrate most of their runoff from high-elevation snowpacks in the Coast, Columbia, and Rocky Mountains have very low sensitivity to MPB-related disturbance and salvage harvest (i.e., Fraser River at Hope, Figure 8b).

Also motivated by the potential for widespread hydrologic changes, Carver et al. (2009) (see also Weiler et al. 2009) developed and applied a peak flow hazard model to assess impacts to third-order catchments within the Fraser River basin. The hazard-based approach presumes that areas that generate more runoff in a watershed during a snowmelt event are more sensitive to land cover change. The approach of Carver a)

b)

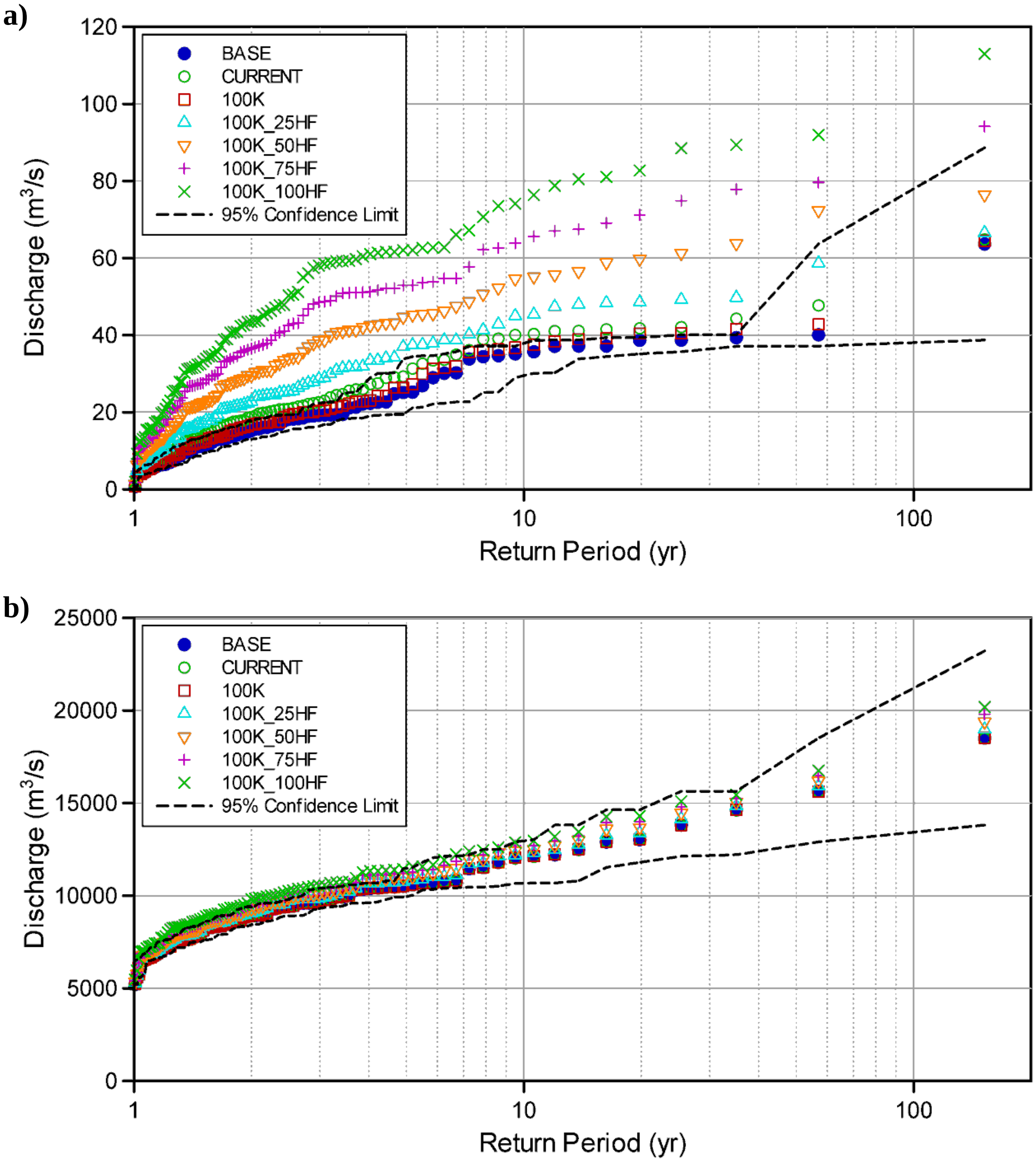


**Figure 8**. Peak flow frequency curves by scenario for a) Baker Creek (1570 km$^2$), and b) the Fraser River at Hope (216,000 km$^2$). Scenarios correspond to: 1995 forest cover (BASE); cumulative beetle-kill and harvesting from 1995 to 2007 (CURRENT); 1995 plus 100% mortality of mature pine (100K); 100K plus 25% salvage harvesting (100K_25HF); 100K plus 50% salvage harvesting (100K_50HF); 100K plus 75% salvage harvesting (100K_75HF); and 100K plus 100% salvage harvesting (100K_100HF). The 95% confidence region for the baseline scenario is shown by the dashed lines. Redrawn from Schnorbus et al. (2010).

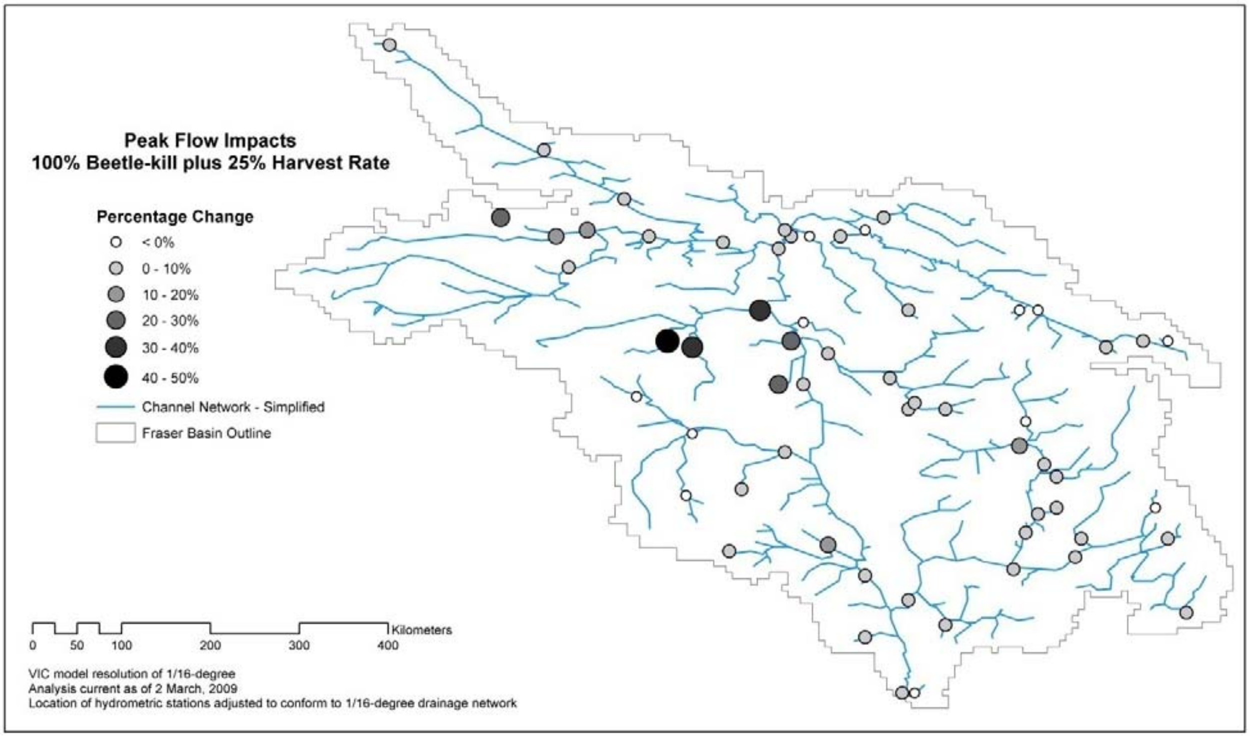


**Figure 9**. Percentage change of the 20-year peak flow event from baseline (1995 forest cover) by sub-basin locations for the hypothetical scenario projecting 100% kill of mature lodgepole pine plus 25% clearcut salvage harvest (by area) of dead pine (scenario 100K_25HF in Figure 8). Redrawn from Schnorbus (2009).

et al. (2009) is to model runoff by mapping dominant runoff source areas, which are based on several different runoff-producing mechanisms, across the landscape and then to prescribe different runoff contributions for each mechanism during peak snowmelt. Carver et al. (2009) applied their approach to eight watersheds (the Stuart, Nautley, Salmon, Chilcotin, Quesnel, Cottonwood, and Yalakom Rivers and Spius Creek), whereby they quantified projected changes in the mean annual peak flow for several different scenarios (which are qualitatively similar to those of Schnorbus et al. [2010]). Again, model results suggest that peak flow increases following forest disturbance, that salvage harvesting has a larger impact than beetle kill, and that peak flow change increases with increasing disturbance area.

Seven watersheds examined by Carver et al. (2009) were also examined by Schnorbus et al. (2010), and the combined results for two similar scenarios are shown in Figure 10.. Although it can be argued that both models agree on the relative sensitivity of the seven sub-basins to a given disturbance (particularly salvage harvesting), it is clear that the magnitude of change is very different between the models. Schnorbus et al. (2010) show a substantially smaller peak flow change for a predominantly beetle-kill disturbance (Figure 10a), but a larger change for the salvage harvesting scenario (Figure 10b). These discrepancies highlight differences in model paradigm and structure, experimental approach, scenarios (i.e., differences in pine extent and disturbed forest area), and assumptions regarding beetle-kill effects on melt rates. For instance, Schnorbus et al. (2010) generate a 91-year ranked sample of annual maximum peak flow events (by running the VIC model for 91 years for each scenario at a daily timestep), from which the median of the sample (the two-year event) is used to estimate the average peak flow event. Carver et al. (2009) use 30-year averages of daily temperature and precipitation to generate a single

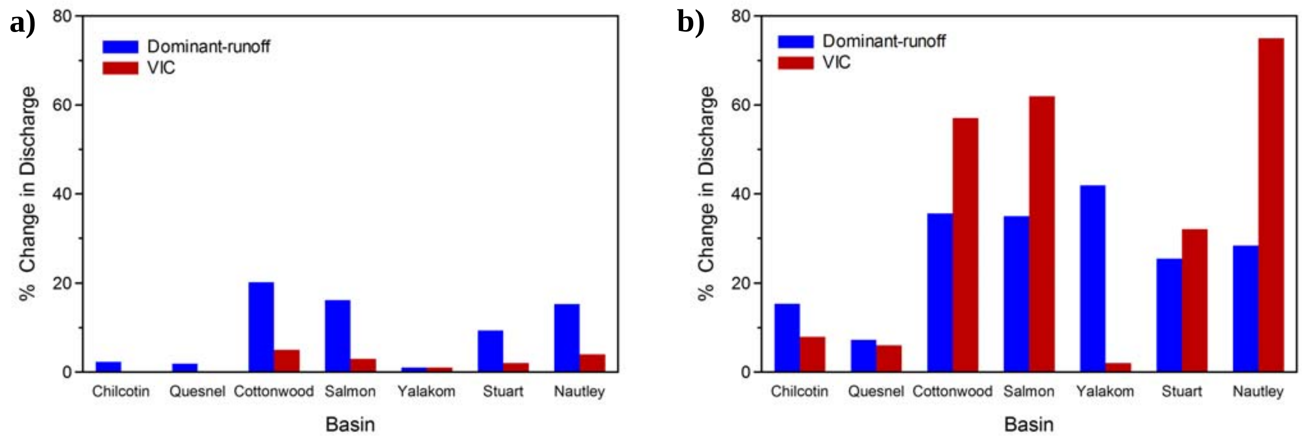


**Figure 10**. Relative change in mean annual peak flow as simulated using the VIC model (*T* = 2 years; Schnorbus et al. 2010) and the dominant-runoff model (*T*=2.33 years; Carver et al. 2009) for scenarios comparing 1995 forest cover to: a) 2007 forest cover, and b) 100% salvage harvest of all mature pine for seven Fraser River sub-basins.

peak flow event which is assumed to be the average peak flow event. Further, Schnorbus et al. (2010) use differences in leaf area index[6] (LAI) between healthy and dead pine stands to parameterize the melt rate, whereas Carver et al. (2009) assume that the melt rate in grey stands is two-thirds that of a healthy stand. Nevertheless, the results for beetle kill from Schnorbus et al. (2010) may be conservative, reflecting only the impact of forest cover changes (based on *LAI*) immediately following mortality (i.e., in the first 10 years following attack; see Section 2.1.3).

# 3. Discussion

## 3.1 Consequences and Implications

Snow accumulation is strongly affected by forest canopy interception and, on average, tends to increase with stand disturbance where forested sites have less snow than beetle-killed sites, which have less snow than forest openings. Nevertheless, results for snow accumulation are highly variable in space and time, and some studies report higher snow accumulation in the forest than in the open. Further, forest/grey/open ratios of snow accumulation appear to be site specific (i.e., are closely related to climate, elevation, stand type, etc.), with no consistent landscape-scale relationship to canopy density parameters. It is possible that much of the variability in observed forest/open ratios derives from inherent limitations in snow accumulation studies (Winkler and Boon 2009). Results comparing beetle-killed and clearcut forest stands may be confounded by the fact that climate and topography may have a more dominant influence on snow accumulation and melt than forest cover does (Alila et al. 2009; Jost et al. 2007). Regardless, the general implication is that beetle kill results in snow accumulation that is intermediate between that of a healthy forest and that of a clearcut (or forest opening).

Snow ablation shows a much more consistent response to stand disturbance, with average ablation rates increasing from healthy forest stands to dead (or grey) stands to clearcuts and openings. Peak ablation

[6] *Leaf area index* is total leaf area per unit ground area (typically measured in units of $m^2/m^2$).

rates are similar in forested (healthy and dead) stands, but are substantially higher in openings. Changes in melt or ablation rates are a function of changes in the energy balance, where snowmelt energy increases with decreasing forest cover, predominantly due to increases in solar radiation and sensible heat. Results suggest that snow ablation or melt rate in a beetle-killed stand is between that of healthy forests and openings. Furthermore, changes in snow ablation rates may be predicted based on measuring changes in canopy cover parameters that relate directly to solar radiation transmittance (e.g., Teti 2009).

At the stand scale, MPB disturbance reduces interception loss and overstorey transpiration, resulting in higher snow accumulation and rainfall on the ground and higher soil moisture during the spring, summer, and fall. For well-drained sites, this results in increased summer and fall drainage, and we can surmise that this leads to increased local annual and seasonal streamflow. Higher snowmelt rates combined with higher soil moisture in the spring can result in higher episodic soil drainage, potentially generating increases in local freshet and peak runoff. However, such effects will likely be quite sensitive to soil moisture deficits carried over from the previous summer and fall. For areas that exhibit poor drainage characteristics (due to poorly drained soil, a receiving slope position, and/or low drainage density), increased soil moisture storage is favoured over drainage, resulting in increased water table heights and possibly saturated soils. This has led to a loss of operational “summer ground” in some areas of the Vanderhoof Forest District (Rex and Dubé 2009). The relative response of the stand water balance from beetle kill versus salvage harvesting is unclear. Model results suggest that, on average, soil drainage can be similar between red, grey, and clearcut stands (Spittlehouse 2007). Other evidence (empirical and model-based) suggests that interception and/or evaporation from the understorey and litter layers can compensate, to varying degrees, for the loss of overstorey canopy in beetle-killed stands (Carlyle-Moses 2007; Silins et al. 2007; Stednick and Jensen 2007).

Over time, the hydrologic effects of beetle kill relative to salvage harvesting will be a function of both the degeneration rate of dead standing pine and the recovery and regeneration rate of any secondary structure in unsalvaged stands compared to the recovery rate of planted seedlings in salvaged stands. Generally, the short-term (less than 20–25 years since disturbance) hydrologic impact will almost always be higher in salvage-harvested stands than in unsalvaged, beetle-killed stands. However, over the long term (more than 25–30 years since disturbance) planted clearcuts and partially harvested (and planted) stands may have higher recovery potential than unsalvaged stands (Lewis and Huggard 2010; Teti 2009). Clearly, the relative short-term and long-term impacts of beetle kill versus clearcut harvesting will depend upon a number of factors. These include: 1) the attack severity (i.e., the proportion of the stand killed), which will depend upon the proportion of susceptible pine in the overstorey; 2) the presence, amount, and type of secondary structure; 3) the release and regeneration delay of residual saplings and seedlings; and 4) the comparative growth and survival rates of natural regeneration and planted seedlings (which depends on site index and other environmental conditions). The implication is that hydrologic impacts and recovery rates will be site and watershed specific.

Consistent with the stand-scale results already discussed, the empirical evidence indicates that, at the watershed scale (drainage areas ranging from 133 $km^2$ to about 2000 $km^2$), biotic forest disturbance results in increased annual, spring, and summer/fall water yields (Uunila et al. 2006, and references therein). Recent empirical data, however, suggests that the water yield response to beetle kill can be variable (changes range from > 30% to -20%), and is strongly related to the presence and amount of secondary structure in the affected watersheds (Stednick and Jensen 2007). Following beetle kill, an absence of secondary structure results in water yield increases whereas an abundance of secondary structure can result in water yield decreases. Changes in annual peak flow, however, are inconclusive and/or insignificant in all studies.

Given the challenges of conducting watershed-scale empirical research on the effects of large-scale beetle kill (or forest disturbance in general), recent understanding of the impacts of the latest widespread beetle epidemic derives predominantly from modelling studies. These allow the effects of beetle kill and clearcut salvage harvesting to be separated. Unfortunately, some modelling studies only report on the effects of salvage harvesting, or the combined effect of both beetle kill and salvage harvesting. A related issue is that modelling studies often use different forest cover and disturbance scenarios, making comparison of results difficult or impossible.

Qualitatively, modelling results are generally consistent with observed effects on annual and seasonal water yield. The impact of salvage harvesting is expected to increase annual water yield, with the magnitude of change correlated to the relative area of the watershed affected. The cumulative impact of beetle kill and salvage harvesting is also projected to increase freshet water yield (April through July), where the magnitude of change is correlated with the combined area of beetle kill and salvage harvest. In contrast to the empirical data, modelling studies predict a consistent and sometimes dramatic response of annual peak flows to beetle kill and salvage harvesting. Both beetle kill and salvage harvesting have the potential to affect the peak flow regime, where the impact is correlated to both disturbance type and area affected. When the impact is sufficiently large, the peak flow regime is adjusted such that for a given frequency of occurrence the magnitude is increased, and for a given magnitude the event becomes more frequent. Combined results suggest that beetle kill has a smaller impact on peak flow than the cumulative effect of both beetle kill and clearcut salvage harvesting. Results also indicate that peak flow increases from baseline with increasing area disturbed. However, where study watersheds and scenarios overlap, the various modelling studies do not necessarily agree on either the magnitude of change for a given disturbance or on the relative sensitivity of the peak flow regime to different disturbance types (i.e., beetle kill versus salvage harvesting).

Modelling of beetle-related disturbance in the Fraser River basin provides an assessment of hydrologic impacts at a range of watershed scales from a regional to a macro scale (from 330 $km^2$ to over 217 000 $km^2$). Results show that the potential beetle-related forest impact upon peak flow is a function of both the distribution of lodgepole pine as well as the spatial variation of relative runoff production at the landscape scale. Although lodgepole pine has wide climatic and elevation amplitude, its distribution in the Fraser River basin is mainly confined to the relatively dry Interior Plateau; it is generally absent from the mid- and high-elevation areas of the Coast, Columbia, and Rocky Mountains (BCMoFR 2010). However, within the Fraser River basin most runoff is derived from snowmelt in the alpine and sub-alpine regions of the Coast, Columbia, and Rocky Mountains. The general conclusion is that small to moderately sized watersheds located on the Interior Plateau with a preponderance of area covered by lodgepole pine are most sensitive and are at highest risk to beetle-related disturbance. As watershed area increases, more runoff is progressively incorporated from higher-elevation regions of the landscape, which are wetter and contain little to no pine. Consequently, watersheds become increasingly more robust to beetle-related impacts with increasing drainage area. That is, sensitivity is high in watersheds that contain a majority of mature pine and in which most runoff is derived from the pine-covered area. Consequently, disturbance risk is highest in the pine-dominated landscape of the Interior Plateau in small- to medium-scale watersheds (10–1000 $km^2$; e.g., Nautley River and Baker Creek); intermediate for meso-scale watersheds (1000–100 000 $km^2$; e.g., Quesnel and Nechako Rivers), and low for macro-scale watersheds (over 100 000 $km^2$; e.g., Thompson River and Fraser River at Hope). When cautiously extrapolating these results, the implication is that, despite lodgepole pine's widespread distribution in British Columbia, peak flow impacts will be highest at the local and regional scales (less than 1000 $km^2$). However, where peak flow changes occur they can be substantial, with possible attendant consequences for channel morphology, aquatic habitat, infrastructure survival, and public safety.

## 3.2 Outstanding Knowledge Gaps and Research Requirements

In order to improve upon our current understanding of the hydrologic consequences of MPB-related disturbance and our ability to manage forested landscapes sustainably from a water resource perspective, several outstanding knowledge gaps discussed below. These include: 1) a retrospective analysis of climate, hydrometric, and forest disturbance data; 2) continued efforts to quantify the impacts of forest cover difference on snow processes; 3) quantification of the effects of secondary structure on hydrologic function; 4) quantification of the transient and dynamic nature of beetle kill and salvage harvesting; 5) validation of the use of ECA to predict disturbance impacts and recovery; 6) continued use of stand and watershed modelling; 7) improvements to hydrologic modelling dealing more explicitly with biotic forest disturbance; and 8) general recommendations for monitoring and data collection. Although significant progress has been made over the last 5 years, many of the research recommendations that have been made previously (e.g., Helié et al. 2005 and Uunila et al. 2006) still apply. Outstanding knowledge gaps and broad research activities suggested for future studies are identified as follows:

1. A retrospective analysis of MPB impacts on streamflow at a watershed and landscape scale using current topographic, soil, and geologic spatial data layers, and time series hydrometric and climate data. Given that the current MPB infestation has now been progressing for well over a decade, there may be sufficient information in existing data sets to use statistical models to quantify the effects of disturbance type and area affected on streamflow. A major component of this research would be constructing a historical annual time series of spatially homogeneous forest cover from roughly 1995 to the present, specifically including changes due to forest management, beetle kill, and wildfire disturbance.
2. The processes of snow accumulation and melt are the dominant components of the hydrologic cycle in the regions of the BC Interior. As such, and given the variability and uncertainty in recent results (e.g., Winkler and Boon 2009), particularly with respect to snow accumulation, continued research is required to quantify the effects of beetle kill and salvage harvesting on these fundamental processes. Building on the work of Alila et al. (2009), Jost et al. (2007), and Winkler and Moore (2006), greater quantitative understanding is needed to convey the effects of forest cover variability on snow accumulation and melt in the context of the variability imposed by climate and topography.
3. The nature of biotic (i.e., non-stand replacing) disturbance is such that the presence and abundance of secondary structure governs the potential magnitude of hydrologic change. Secondary structure is an important element with respect to forest microclimate, snow processes, stand water balance, and streamflow. Therefore, there is a need to quantify the relationship between secondary structure and hydrologic impacts at the stand scale (snow accumulation and melt, interception, evaporation and transpiration, soil moisture, and drainage) and watershed scale (streamflow and water yield); particularly in light of recent evidence (i.e., Stednick and Jensen 2007), which suggests that the presence of secondary structure may result in increased transpiration rates and reduced water yields following beetle kill.
4. It is necessary to quantify the dynamic nature of forest disturbance as it governs hydrologic processes. Research must continue with efforts to relate hydrologic functioning to appropriate measurable and transient forest stand characteristics (e.g., age, height, canopy density, radiation transmittance, canopy closure, species composition, basal area) that deal explicitly with the non-stand-replacing nature of biotic disturbance. It is also necessary to discriminate the effects of stand degeneration and regeneration on each individual component of the water balance (snow accumulation, melt, interception, evaporation, transpiration, soil moisture, and drainage); this accounts for the possibility that deterioration and recovery rates may differ between water balance components.
5. Although the use of equivalent clearcut area (ECA) represents an attempt to describe the transient nature of forest disturbance, it is only an index that has been shown to have poor

predictive ability for quantifying streamflow change at the watershed scale (due to such confounding effects as climate, elevation, slope, aspect, and runoff synchronization). Nevertheless, because it is based on stand characteristics with known relationships to snow accumulation and melt (height and stand density; BCMoF 2001), it may have predictive utility at the stand scale. Therefore, it is necessary to test and validate the use of ECA to describe stand-scale recovery, particularly in unsalvaged, beetle-killed stands. Monitoring stand characteristics and hydrologic fluxes over the long term is also necessary to validate and update current models (i.e., Lewis and Huggard 2010) of stand deterioration and recovery rates for beetle-killed and managed stands.

6. Because of the difficulty in establishing proper before–after/control–impact (BACI) studies, particularly for large areas, water balance and hydrologic modelling remain important means of predicting and projecting MPB-related disturbance from the stand to the landscape scale. Stand water and energy balance modelling is an effective tool for improving our quantitative understanding of the relationship between forest characteristics and hydrologic fluxes and how forest disturbance results in hydrologic change. Hydrologic modelling is an effective means of extrapolating relationships obtained at the stand scale to larger landscapes, where hydrologic response to disturbance is also influenced by variability in such factors as climate, forest structure (e.g., height, density, and canopy cover), tree physiology (e.g., age, transpiration rate), topography (e.g., slope, aspect, relief, and elevation), and basin topology (e.g., drainage density and stream order). Hydrologic modelling is necessary to understand and predict the effects of forest disturbance over large regions and at locations downstream of the disturbance area.
7. Future hydrologic modelling efforts should focus on determining and validating appropriate parameters to deal more explicitly with biotic forest disturbance (e.g., the effects of partial canopy removal and tree mortality on interception loss, and the effect of understorey release on transpiration rates). Future modelling efforts should also explicitly account for the transient nature of the forest landscape following disturbance when making predictions and projections of future streamflow change, particularly the effects of stand degeneration and regeneration over time. Greater emphasis should also be placed on quantifying uncertainty and sensitivity analysis for risk analysis and decision making. Future modelling efforts require access to a standard set of forest disturbance scenarios or projections: this would facilitate better comparison of results from different models. Hydrologic modelling also serves as a tool for establishing and evaluating simple indicators of watershed sensitivity to forest disturbance.
8. In order to continue with any meaningful research into the impacts of MPB-related disturbance on hydrology, we must remain invested in long-term monitoring and data collection. This will require the continuation and maintenance of existing long-term field studies (e.g., Upper Penticton Creek, Baker Creek, Cotton Creek, and Mayson Lake) as well as continued support for hydrometric monitoring. General data and monitoring requirements are:
    a. An inventory of secondary structure in lodgepole pine stands.
    b. Long-term monitoring of stand microclimate and water balance (e.g., snow, interception, soil moisture, and water table) combined with standardized measurements of forest structure (e.g., canopy cover and transmittance) in a range of healthy, beetle-killed, and managed stands.
    c. Long-term hydrometric monitoring (discharge, stream temperature, and sediment) at a range of scales within and downstream of beetle-infested areas.

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