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Effects of Forest Harvesting on Warm-Season Low Flows in the Pacific Northwest: A Review

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Abstract

Paired-catchment studies conducted on small (< 10 km^2) rain-dominated catchments revealed that forest harvesting resulted in a period of increased warm-season low flows ranging from less than five years to more than two decades, consistent with the results of stand-level studies and process considerations. Of the five paired-catchment studies in snow-dominated regions, none revealed a statistically significant change in warm-season low flows in the first decade following harvest, although two exhibited non-significant higher flows in August and September and one had lower flows. Two studies, one of rain-dominated catchments and one of snow-dominated catchments, found that summer low flows became more severe (i.e., lower) about two decades or so following harvest. These longer-term results indicate that indices such as equivalent clearcut area, as currently calculated using monotonic recovery curves, may not accurately reflect the nature of post-harvest changes in low flows. Studies focussed on medium to large catchments (tens to thousands of km^2 in area) found either no statistically significant relations between warm-season low flows and forest disturbance, or inconsistent responses. Attempts to synthesize existing studies are hampered by the lack of a common low-flow metric among studies, as well as detailed information on post-harvest vegetation changes. Further field research and process-based modelling is required to help elucidate the underlying processes leading to the results from these paired-catchment studies and to enhance the ability to predict streamflow responses to forest harvesting, especially in the context of a changing climate.

KEYWORDS *streamflow; forestry; low flows; fish habitat; hydrologic recovery*

Introduction

Over the past few decades, there has been increasing concern about in-stream flow needs, also known as environmental flow needs (EFN), which describe the quantity and timing of water that are necessary to support aquatic ecosystems (Jowett, 1997; Poff & Zimmerman, 2010). In particular, the paradigm of the natural flow regime has become a key perspective for understanding the impact of human activity on aquatic ecosystems (Poff et al., 1997; Richter et al., 1996). Much of the earlier work on EFN focussed on quantifying the effects of dams and diversions, but attention has more recently turned to the effects of changes in land use and land cover (LULC) (Poff et al., 2006). Poff et al. (2006) noted that although the directions of change associated with various land uses are generally well understood, the magnitudes should vary among regions in response to differences in soils, geology, topography, and climatic characteristics, and they stressed the need for further work to understand the geographic variability of hydrological alterations associated with LULC.

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The effects of forestry on streamflow have been investigated for at least a century, with one of the earliest studies being conducted at Wagon Wheel Gap in Colorado (Bates & Henry, 1928; van Haveren, 1988). In previous studies, most emphasis has been on water yield and peak flows, especially for catchments with snow-dominated hydrologic regimes, with less attention paid to low flows (Moore & Wondzell, 2005). Over the last decade, however, there has been increasing concern about forestry effects on low flows, particularly in the context of climatic change, which is projected to result in more frequent and severe droughts in many regions, including the Pacific Northwest (Pike et al., 2010).

The objective of this article is to summarize and, to the extent possible, synthesize published research that relates to the effects of forestry operations on low flows during summer and early autumn. The review focusses on studies in the Pacific Northwest, broadly defined as the area from southern Alaska to northern California that drains into the Pacific Ocean. However, studies in Alberta, Wyoming, Colorado, and the Sierra Nevada were included as they are relevant to understanding the effect of harvesting on low flows in snow-dominated regimes. The article builds on previous reviews by Johnson (1998), Pike and Scherer (2003, 2004), Moore and Wondzell (2005), and Winkler et al. (2010a, 2010b). It complements two recent reviews on the effects of forestry on streamflow by Goeking and Tarboton (2020) and Coble et al. (2020) by critically evaluating the results of previous studies, particularly in relation to the range of low-flow metrics used, data quality, and interpretation of statistical significance.

Hydroclimatic Context of Low Flows and Sensitivity to Forestry

Streamflow regimes in the Pacific Northwest can be broadly grouped into four classes: rain-dominated (or pluvial), snow-dominated (or nival), hybrid rain-snow, and pro-glacial (Eaton and Moore, 2010). The processes that control summer-autumn low flows differ among the regimes, and hence the sensitivity of summer-autumn low flows to forestry operations should also differ.

Rain-dominated regimes are found at low- to mid-elevation sites in coastal areas and generally coincide with the Coastal Douglas-fir (CDF) and Coastal Western Hemlock (CWH) biogeoclimatic zones within the British Columbia Biogeoclimatic Ecological Classification (BEC) zone system (Trubilowicz et al., 2013). These regimes experience variable but generally high flows during the wet, cool winters. Low flows occur through summer and early autumn during extended spells of dry, warm weather. Low flows could also occur during extended dry periods during winter, but these do not appear to have been considered in forestry-related studies.

Snow-dominated regimes are ubiquitous in the British Columbia Interior and across the Rocky Mountains into Alberta, except for some low-elevation valleys in the southern part of the region. In these regimes, most winter precipitation falls as snow and is stored until spring. Snow-dominated regimes are also found at higher elevations in the Coast and Cascade mountains, typically coinciding with the Mountain Hemlock (MH) BEC zone in British Columbia and equivalent zones in other jurisdictions. The highest flows generally occur during the snowmelt freshet, usually extending into May or June, and are often supplemented by rain falling on a melting snowpack or on moist soils for a period after the snowpack disappears. Flows then typically decline through summer and autumn until late September or October, when autumn rains, in combination with reduced evapotranspiration, can cause an increase in flow. Once temperatures drop below 0°C for the season, there is generally an extended baseflow recession that extends to late winter or early spring, when the freshet begins. Thus, extreme low flows can occur in summer-autumn or in late winter-early spring.

Snow dynamics differ between the warmer, maritime snow zones in the Coast and Cascade mountains and the colder, drier snow zones in the Interior mountains and plateaus (Haegeli & McClung, 2007; Mock & Birkeland, 2000). An important difference is that the maritime zones can receive an order of magnitude more precipitation (Moore et al., 2010), and they typically experience above-freezing temperatures and rain-on-snow events one or more times each winter (Trubilowicz & Moore, 2017). These differences in snow climate influence the contrast in snow dynamics between forest and open sites (Lundquist et al., 2013). For example, in drier regions such as the Okanagan Plateau in British Columbia, snow tends to disappear earlier in openings than under forest (Winkler et al., 2015), whereas snow lasts longer in openings in the more humid continental-maritime setting of Idaho (Hubbart et al., 2015).

Hybrid regimes are common in catchments in the Coast and Cascade mountains that span a range of elevations—e.g., with CWH in the lower elevation zones and MH in the higher zones (Trubilowicz et al., 2013). The lower elevations tend toward a rain-dominated regime, whereas the upper elevation zones tend toward a snow-dominated regime. Because of the winter rain contributions, late-winter low flows are not as severe or frequent as in snow-dominated regimes. For these regimes, the main low-flow period of concern occurs in summer-autumn.

Pro-glacial regimes occur in catchments with more than about 2–5% glacier cover (Eaton & Moore, 2010; Stahl & Moore, 2006). These regimes are similar to snow-dominated regimes, but the snowmelt freshet typically extends into June and July due to the generally higher elevations in glacierized catchments. During August and September, following the main pulse of snowmelt runoff, glacier melt becomes a significant contributor to streamflow, especially during extended periods of warm, dry weather. As a consequence, summer-autumn low flows in glacier-fed streams tend to be higher than those for nival regimes. Therefore, this review will focus on streams with pluvial, nival, and hybrid regimes.

Influence of forestry operations on processes controlling low flows

Overview

During extended periods of dry weather, streamflow in unglacierized catchments is sustained by the depletion of water stored within a catchment as surface water (wetlands, ponds, lakes) and subsurface water (soil moisture and groundwater). For wetlands and small ponds, it is conceivable that forest harvesting could influence evaporative losses from the surface by reducing shading and increasing wind speed over the water, which would increase evaporation and thus reduce flow downstream of the wetland or pond. For example, Guenther et al. (2012) showed that evaporation from a headwater stream increased following a thinning treatment. However, the influence of harvesting on pond evaporation and low flows does not appear to have been quantified by any published literature. This issue should be considered further. For lakes that are large enough to experience minimal shading by lakeside forest prior to harvest, the main effect of forestry would be to modify the amount and timing of inflow to the lake via surface and subsurface pathways.

The discharge of water from subsurface storage to the stream network is a function of the volume of storage and its spatial distribution within a catchment (Moore, 1997), and it is controlled by topography, soil characteristics, and surficial and bedrock geology (Tague & Grant, 2004). For example, lower permeability bedrock geology and shallow soils generally lead to lower low flows compared to catchments with deeper soil and higher permeability bedrock (Johnson, 1998). Forestry practices can influence subsurface storage dynamics and discharge by modifying inputs to storage via precipitation and snowmelt and/or outputs by interception loss, soil evaporation, and transpiration. In some geological contexts, it may be possible for groundwater flow across catchment boundaries to contribute to streamflow in a catchment of interest. In such a case, groundwater contributions to low flows may not be influenced by changes in LULC within the catchment.

Rainfall interception loss

Most rain falling onto a canopy will be intercepted by foliage, and some of it will be lost to the atmosphere by evaporation. The remainder reaches the ground primarily as throughfall, with a smaller amount becoming stemflow (Rothacher, 1963; Spittlehouse, 1998). For small storms with rainfall depths similar to the interception capacity of the canopy layer, nearly all rain may be intercepted and lost, particularly from the epiphyte-laden canopies of old-growth forests (Pypker et al., 2006). The fraction of rainfall that is lost to evaporation decreases as storm magnitude and intensity increase (Rothacher, 1963; Spittlehouse, 1998). At rain-dominated sites, interception loss as a percent of gross precipitation is greater in summer than winter (Rothacher, 1963). On an annual or seasonal basis, interception loss from conifer forests in the Pacific Northwest generally represents about 10–30% of total rainfall, depending on canopy characteristics and climatic conditions (e.g., Beaudry & Sagar, 1995; Link et al., 2004; Rothacher, 1963; Sollins et al., 1980; Spittlehouse, 1998).

In coastal areas, fog or clouds can condense onto a forest canopy and then drip to the forest floor. Fog and cloud drip may significantly augment precipitation in coastal and some high mountain catchments. For example, Harr (1982) found that rainfall reaching the ground in a clearing was 30% lower than throughfall measured below an adjacent forest stand in the Oregon Coast Range. In terms of its significance for low flows, an important consideration is the seasonal timing of fog drip, as discussed in more detail in the section titled "results for medium to large catchments."

Snow dynamics

Two aspects of snow dynamics are potentially of consequence for low flows: the total volume of snowmelt, which is a first-order control on the seasonal recharge to various storage reservoirs within a catchment, and the timing of snow disappearance, which controls the start of the seasonal recession toward late-summer baseflow conditions. Past studies provide consistent evidence that the total volume increases following forest harvesting, whereas the effect on snow disappearance is more variable.

Snow accumulation tends to be higher in openings than under forest canopies in both maritime and interior sites, with clearcuts typically accumulating about 30–50% more snow, although differences of over 100% have been recorded (e.g., Storck et al., 2002; Toews & Gluns, 1986; Troendle & King, 1987; Winkler & Boon, 2005). In colder regions, much of the difference between open and forest sites is the result of the loss of snow intercepted in the forest canopy. At warmer sites that experience frequent above-freezing air temperatures, most of the intercepted snow either sloughs off as clumps or melts and then drips from the canopy (Storck et al., 2002).

It has been suggested that, beyond a clearing size of about 15 tree heights (15H) in linear dimension, snow accumulation could decrease relative to adjacent forest due to increased wind speeds, which would increase evaporation/sublimation losses and transport of snow out of the clearing into adjacent forest (Swanson, 1980; Troendle & Leaf, 1980). However, Toews and Gluns (1986) reported that snow accumulation was consistently enhanced in clearings up to 42H in size.

The removal or reduction of forest canopy through harvesting allows more solar radiation to reach the snow surface and also results in higher wind speeds below the canopy, which can result in greater energy exchanges associated with sensible and latent heat transfer. As a result, snowmelt rates are typically 30% to over 100% higher in the open (e.g., Berris & Harr, 1987; Toews & Gluns, 1986; Winkler et al., 2005).

In contrast to the consistent results concerning total volume, the difference in the timing of snow disappearance between open and forested sites can be quite variable. In the southern Interior of British Columbia, snow generally disappears one to two weeks earlier at open sites than under mature forest, reflecting the fact that the higher melt rates in openings are more than sufficient to offset the effect of greater accumulation (e.g., Toews & Gluns, 1986; Winkler et al, 2005; Winkler et al., 2015). This advance in the timing of snow disappearance was also observed at south-facing, mid-elevation sites (ca. 1000 m above sea level) in the southern Coast Mountains of British Columbia, where over 250 mm water equivalent of snow remained under mature forest (relative to a peak of about 400 mm water equivalent) at the time snow disappeared from a clearcut. However, snow disappearance at north-facing sites was roughly synchronous among clearcut, mature forest, and two regenerating stands (Hudson, 2000). Dickerson-Lange et al. (2017) found that, for 12 out of 14 sites in the U.S. Pacific Northwest with coniferous forest and maritime and transitional maritime-continental climates, snow disappearance was either synchronous at paired open-forest plots, or up to 13 weeks earlier under forest canopy. Snow disappeared earlier in the opening at only two sites that generally experienced high winds, which influenced snow accumulation. Forest density and the size of openings also influence the timing of snow disappearance (Lundquist et al., 2013). More research is required to clarify the reasons for this variability of snow disappearance timing among sites, and to develop approaches to represent this variability within process-based simulation models.

Transpiration

Transpiration removes water from soil moisture storage, which can create more extreme low flows through two mechanisms. First, the removal of water reduces the amount of subsurface water that would potentially flow downslope and discharge to the stream channel. Second, when soil moisture is drawn down below field capacity, an increasing amount of any rainfall will be used to recharge soil moisture storage and would thus be unavailable to contribute to streamflow. Indeed, studies of hillslope hydrology in many regions have identified storage thresholds below which rainfall produces minimal or no streamflow response (e.g., Graham et al., 2010).

Water extraction by transpiration influences streamflow at different time and space scales. For example, transpiration in upslope areas reduces the amount of soil moisture that flows downslope toward the stream channel; the effect on streamflow would be expressed on a multi-day-to-seasonal time scale. However, transpiration from the riparian zone can have a rapid effect on streamflow, in many cases causing drops in streamflow during the day, with recovery in the evening (Bond et al., 2002; Moore et al., 2011).

Over the last decade or so, thinning has become a common treatment in the U.S. Pacific Northwest, especially in densely stocked second-growth Douglas-fir stands, to increase habitat diversity (e.g., Anderson & Poage, 2014). Thinning is also being used in forest stands surrounding urban areas to reduce fire risk, for example, near Whistler, British Columbia (Barde, 2018). In addition, some variable retention treatments resemble thinning even if their ultimate silvicultural objectives may differ (Beese et al., 2003). Bladon et al. (2006) found that total daily transpiration for individual trees was greater after a variable retention treatment (the retention of 10% of the basal area) than at a control site for white spruce (*Picea glauca*) and paper birch (*Betula papyrifera*), but not for balsam poplar (*Populus balsamifera*). However, they inferred that total stand transpiration was reduced and, in combination with increased rain reaching the soil surface, resulted in higher soil moisture than in control sites.

Roads

Roads and their drainage systems have three main effects on streamflow generation: the enhanced generation of infiltration-excess overland flow over relatively impervious road surfaces, the interception of subsurface flow and conversion to surface flow in ditches, and the routing of road surface runoff and intercepted subsurface flow to the stream channel via drainage structures (ditches, culverts, etc.) (Coe, 2004). The diversion of road surface runoff and intercepted subsurface flow to drainage structures and thus to the stream channel would result in the reduced recharge of soil moisture and groundwater in areas downslope of these disturbances, which would act to reduce the amount of water potentially available for baseflow. However, the subsurface flow captured by a ditch may be directed onto a slope below a drainage relief culvert and re-infiltrate to become subsurface flow (Wemple & Jones, 2003).

The net effect of roads should be to reduce baseflow contributions to late summer streamflow. However, a road network that is well connected to the stream channel could conceivably increase streamflow contributions from summer stormflow during rain events. The magnitude of the effect of roads would depend on their layout, design, and construction, especially in relation to the connectivity of surface flow paths between roads, ditches, and stream channels. The effect of roads could conceivably depend on weather conditions, with a negative effect for extended dry spells due to the reduction in baseflow contributions and a positive effect for periods with rainfall due to increased stormflow production.

In landslide-prone terrain, road-related failures that introduce significant quantities of debris into a stream channel can indirectly affect low flows because the impacted stream channel can experience channel widening, the loss of roughness, and the loss of pool depth, which all effectively reduce the quantity of available habitat during the late summer low-flow season for any given discharge (Reid et al., 2020). For example, aggradation following inputs of coarse sediment could result in an increase in the amount of discharge flowing through the channel bed and a reduction in surface flow.

Soil moisture

At least for the first few years following harvest, soil moisture should tend to be higher during summer and early autumn in comparison to pre-logging conditions due to the reduction in interception loss and transpiration, especially in cases where broadcast burning of slash removes herbaceous and shrubby vegetation (Adams et al. 1991; McNabb et al. 1989). Adams et al. (1991) found that this increase in soil moisture lasted for about four years at a clearcut site in the Oregon Cascades, after which soil moisture was lower than expected, following the establishment of herbaceous, shrubby, and tree species. Ziemer (1964) found that moisture loss in clearings was initially confined to the surface layer due to the lack of rooted vegetation, and that the depth of soil moisture depletion increased with the age of the opening in association with the growth of the regenerating forest.

Hydrologic recovery

Following forest harvesting, the establishment and development of vegetation will influence hydrologic processes and eventually reduce the magnitude of harvesting-related impacts. The trajectory of recovery will depend on the types of vegetation and their rates of growth and successional processes (Jones & Post, 2004). Relatively few stand-level studies relevant to hydrologic recovery have been published, but they do provide useful insights for interpreting the results of catchment-scale studies. A sample of studies is reviewed below.

Hudson (2000) quantified recovery in peak seasonal snow accumulation and post-peak snow ablation rate relative to an old-growth stand in the snow-dominated zone of the southern Coast Mountains of British Columbia. The post-harvest stands had elevations averaging about 1000 masl and were naturally regenerated, consisting of a mixture of subalpine fir, western hemlock, mountain hemlock, western redcedar, and yellow-cedar. Based on a curve fit to the hydrologic recovery for each stand, tree heights of 4, 6, and 8 m were associated with 53%, 75%, and 83% recovery, respectively. At a site on the Interior Plateau of British Columbia, Winkler et al. (2005) found that 15-year-old pine stands exhibited only partial recovery in terms of both peak snow accumulation and melt rate when compared to a fresh clearcut and a mature forest stand. Winkler and Boon (2015) found that tree heights of 5, 10, and 15 m were associated with approximately 10%, 60%, and 90% recovery, respectively. Comparing the results of Hudson (2000) and Winkler and Boon (2015), it is clear that rates and trajectories of hydrologic recovery for snow processes can vary substantially from site to site, likely as a result of both differences in hydroclimate and forest stand characteristics.

Spittlehouse (unpublished) measured interception loss in second-growth stands in the low-elevation rain-dominated zone at Carnation Creek, British Columbia, located on the west coast of Vancouver Island. Forest stands there are dominated by coastal western hemlock, western red cedar, Douglas-fir, and Sitka spruce. The data suggested hydrologic recovery of 53% and 73% for stand ages of 15–20 and 30–35 years, respectively. Based on these stand-level studies, it can be inferred that forestry influences on interception loss and snow deposition may persist for decades following harvest.

Immediately following clearcut harvesting, transpiration rates should decrease due to the loss of deep-rooted plants that can draw upon soil moisture through a relatively extensive rooting zone. As the forest regenerates, transpiration should increase and, at least in some cases, could become greater than for the original stand. In the Oregon Cascades, Moore et al. (2004) found that transpiration varied with species and tree age within riparian forest stands. Young Douglas-fir (ca 40 years old) had greater transpiration rates than old Douglas-fir (ca 450 years old). Alder had greater transpiration than young Douglas-fir, and old Douglas-fir had greater transpiration than old western hemlock. For a Ponderosa pine forest in the Oregon interior, Ryan et al. (2000) estimated that over the period from June 1–August 31, a 40-year old forest and a 260-year old forest that were assumed to have equivalent leaf area indices transpired 170 mm and 117 mm of water, respectively.

Ziemer (1964) mapped soil moisture patterns across 10 clearings that had been logged between one and 12 years earlier in the seasonal snow zone of the Sierra Nevada in California. In the one-year-old opening, moisture loss was restricted to the surface layer due to the lack of rooted vegetation, and the soil moisture content in the upper 1.2 m of soil was 0.14 m^3 m^{-3} higher than in the adjacent forest. A regression between the magnitude of this moisture savings (relative to the adjacent forest) against clearing age suggested that the effect would disappear by 16 years following harvest. The difference in persistence of the logging effect between Ziemer's (1964) study and the four-year duration reported by Adams et al. (1991) likely reflects differences in growth rates between the lower-elevation site in the Cascades (500 to 550 masl) and the higher-elevation sites in the Sierra Nevada (1830 to 2130 masl).

Expected response of low flows based on process considerations

Rain-dominated regimes

In rain-dominated regimes, forest cover has two key influences on summer-autumn low flows: the reduction of rainfall reaching the ground by interception loss, and the extraction of soil moisture for transpiration. Immediately following harvest, both interception loss and transpiration would be reduced, leading to the greater availability of water to contribute to streamflow. Therefore, it is reasonable to expect that summer-autumn low flows would become less severe following harvest. However, as forest regenerates, interception loss and transpiration would increase, reducing the amount of water available to contribute to streamflow through time. As summarized in the section titled "transpiration," immature stands may have greater transpiration loss than the old-growth stands they replaced, leading to more severe summer-autumn low flows compared to pre-harvest conditions. The conversion from species such as western hemlock to Douglas-fir, which has higher transpiration rates, can add to this effect.

Snow-dominated regimes

Two primary mechanisms could influence the response of summer streamflow to forest harvesting in snow-dominated catchments. The first relates to changes in the timing and magnitude of snowmelt. In cases where harvesting advances the date of snow disappearance by one to two weeks, one possible response would be an earlier shift to summer baseflow conditions, resulting in an extended period of flow recession and thus lower late-summer flow. However, this effect could potentially be mitigated by catchment storage because snow accumulation tends to increase following harvest. That is, even though snow may disappear earlier, a greater amount of snowmelt could enter storage as soil moisture, groundwater, or wetlands and then be released gradually over the summer, acting to maintain flows through the recession period. In cases in which snow disappearance is either synchronous between open and forest sites, or is delayed in openings, the increase in snowmelt volume should tend to produce higher flow in late summer-autumn drought periods.

The second mechanism relates to changes in evapotranspiration. As is the case in rain-dominated regimes, both interception loss of rainfall and transpiration would be reduced immediately following harvest, thus tending to result in higher summer-autumn baseflow. However, this effect is likely to be transient, as the magnitudes of interception loss and transpiration would increase as the forest regenerates. Forest re-growth would be slower in snow-dominated zones due to the lower air temperatures and shorter growing season. Hence, the duration of enhanced baseflow following harvest may last longer than for rain-dominated catchments.

Scale effects

The mechanistic hypotheses developed in the preceding three paragraphs can be most clearly applied to smaller catchments that undergo harvesting operations in a relatively short period of time. In larger catchments, the effect of forestry on low flows is more complex. Larger catchments typically include a broader range of storage reservoirs, including lakes and deeper aquifers, that modify the timing of water delivery to the catchment over seasonal to inter-annual time scales. In addition to the variation due to the integrating characteristics of longer-time-scale storage, the influence of cutblocks being harvested over extended periods of time (e.g., patch cuts separated in time by years or decades) causes further complexity. At any given time, a typical medium-to-large catchment in the Pacific Northwest comprises a mosaic of harvest units and naturally disturbed areas that would be in various stages of post-disturbance response. It is conceivable that recently harvested units may be generating more late-summer streamflow than they were under pre-harvest conditions, and some may have transitioned to a state of generating less. Extensive aquifers could act as filters that smooth the spatial and temporal variability of harvesting effects, such that it may be difficult to detect any change and even more difficult to attribute it to harvesting.

Methodological considerations

Analytical approaches

Two broad types of empirical approaches have been used to estimate the effect of forest harvesting on streamflow. The most statistically rigorous is a paired-catchment experiment, which includes at least one untreated control and both pre- and post-harvest data (Hewlett, 1982). The data are usually analyzed using regression analysis and analysis of covariance (ANCOVA). In some cases, it is possible to conduct a paired-catchment analysis using operational streamflow data (e.g., Bowling et al., 2000; Cheng, 1989). The success of paired-catchment analyses depends on how similar the control and treatment catchments are in terms of climate, geology, soils, topography, and vegetation. A major challenge is that the treatment effect will depend on the details of the forestry operations, catchment characteristics, the sequence of weather events that occur in the pre-treatment and post-treatment periods (especially the occurrence of rare, high-magnitude events), and vegetation regrowth. Consequently, it can be difficult to generalize the results to other catchments or operational treatments. A practical challenge is that conducting such experiments requires a significant long-term commitment of funds and effort, or the fortuitous availability of operationally monitored catchments with appropriate histories of forestry activity.

One potential complication with the interpretation of paired-catchment studies over extended post-harvest periods relates to whether the forest cover in the control is old-growth or second-growth. For example, forest cover at Caspar Creek in northern California was dominated by second-growth stands following a logging cycle that was completed by the 1890s, whereas the Upper Penticton Creek catchments in British Columbia were covered by original forest that developed following a forest fire, and the harvesting treatments were first-pass. For catchments covered by old-growth forests, the hydrologic functioning should remain stable over periods of decades, whereas the hydrological functioning of second- (or later) growth forests could drift over periods of decades, depending on the age of the stand at the time the treatment catchment was harvested.

An alternative to paired-catchment experiments is to conduct a time-series analysis of operational streamflow data for a catchment that has experienced known changes in forest cover through time. Statistical techniques such as trend analysis and intervention analysis can be applied to assess the nature and significance of streamflow changes during and following forestry activity (e.g., Zhang & Wei, 2014). Climatic indices can be employed as co-variates—e.g., in a double-mass analysis or ANCOVA—to help control for the effects of climatic variations on streamflow (e.g., Eaton et al., 2010; Zhang & Wei, 2012).

An alternative to empirical approaches is to use computer simulation models. Popular process-based models for forestry related applications in the Pacific Northwest are the Distributed Hydrology-Soil-Vegetation Model (DHSVM) (Wigmosta et al., 1994) and the Regional Hydro-Ecological Simulation System (RHESSys) (Tague & Band, 2001). These models can account for vertical exchanges within the canopy and soil column as well as lateral flow within the soil, including the effects of the road network on flow routing. A modelling approach is attractive because alternative treatments can be applied to the same catchment.

There are three variations on the use of models for hydrologic change detection. In one, a model is calibrated for a catchment for a period of known land cover, and then run using a range of land-cover scenarios. The difference between each scenario and the base simulation is taken as an estimate of the effect of forest harvesting (e.g., Tague & Band, 2001; Whitaker et al., 2002). This approach is based on the assumption that the model accurately reproduces hydrologic responses to forest cover change that are relevant to the streamflow metric of interest, including the effects of post-disturbance regeneration. A second variation is to calibrate a model for a period of known land cover, which is used as a baseline, and then to simulate the entire time series with the land cover fixed to the base condition. The difference between observed and predicted streamflow is taken to be the effect of land-cover change. Bowling et al. (2000) applied this approach to three catchments in western Washington. A third variation is similar to the preceding. However, rather than using the modelled flows directly as an estimate of streamflow for the baseline land cover condition, they are treated in the same way as a control catchment in a paired-catchment analysis. Zégre et al. (2010) applied this approach.

In this review, results from paired-catchment studies are considered the most definitive. However, results using other methods, with the exception of the first model-based approach, were also considered. The results are considered separately for small catchments (up to about 10 km^2) and medium to large catchments (tens to thousands of km^2).

Low-flow indices

A variety of indices have been used to characterize low-flow response to forestry. Perry and Jones (2017) used water yield for July to September, which is generally dominated by the seasonal recession trend in both rain- and snow-dominated catchments. Keppeler and Ziemer (1990) used the total water yield during the low-flow period, defined based on the period in which the discharge in the control catchment dropped below 28 L s^{-1} rather than a period based on fixed calendar dates. While these indices are useful, particularly in relation to water supply, mean flow or total water yield over periods of three or more months may not capture the severity of hydrological drought during the most extreme low flows or times at which aquatic organisms may be particularly sensitive (e.g., times of migration or spawning).

Some studies used the flow on a single day in each year as a low-flow index. Harris (1977) used the minimum daily flow in August and September, while Keppeler and Ziemer (1990) used the daily flow on the last day of the low-flow period. However, many concerns relate to prolonged periods of low flow. Moore and Scott (2005) and Zhang and Wei (2014) also used the seven-day low flow, a statistic that has its origins in water treatment to manage dilution ratios and biological oxygen demand (Annear et al., 2004), as a low-flow index. However, natural production in aquatic ecosystems is related to longer-duration growing season conditions that have been characterized in Wyoming by the average flow during the six-week period from August 1 to September 15 (Binns, 1982), and in Washington state by the 60-day mean summer low flow (Beecher et al., 2010).

A number of studies examined monthly mean flows or water yields (e.g., Moore & Scott, 2005; Winkler et al., 2017). The lowest summer flows tend to occur in August or September in both rain- and snow-dominated catchments in the Pacific Northwest, depending on the timing of both the end of the snowmelt, the occurrence of summer rainfall events, and the arrival of autumn rains. Hicks et al. (1991) used August water yield as a fisheries-relevant index, as did Surfleet and Skaugset (2013). With the increasing lack of climatic stationarity due to global warming, the timing of the 30-day low-flow window is also expected to vary more from year to year, especially in snow-dominated catchments with early freshets and extended summer low-flow periods, or in regions that experience occasional rainy periods during summer. Therefore, in comparison to average monthly flow, the summer 30-day low-flow statistic should provide a more reliable representation of the low-flow conditions that determine aquatic ecosystem production in any given year.

Four studies used a daily time step for analysis. This approach has the advantage that both the timing and magnitude of streamflow departures from pre-harvest conditions can be identified, which can be useful when assessing the potential for impacts on specific aquatic species by taking into account their life history. Winkler et al. (2017) fit a single regression using all of the daily data for the pre-harvest period. However, their approach did not explicitly address issues with temporal autocorrelation and heteroscedasticity (i.e., the error variance about the regression line scales with discharge). While the analysis of Winkler et al. (2017) appeared to capture the shifts in timing and magnitude during freshet in a robust manner, it did not have sufficient sensitivity to detect changes in low flows in late summer and early autumn. Gronsdahl et al. (2019), Perry and Jones (2017), and Segura et al. (2020) conducted paired-catchment analyses on a day-of-year basis. In this approach, streamflow data are grouped into subsets by the day of the year, and the analysis is conducted on each subset. Because sequential values in each subset are separated by a full year, temporal autocorrelation should not be an issue. Also, because of the strong seasonality of streamflow in snow-dominated catchments, stratifying by the day of the year helps to address heteroscedasticity. Gronsdahl et al. (2019), Perry and Jones (2017), and Segura et al. (2020) all applied a logarithmic transformation to address heteroscedasticity and the lack of normality of the error terms. This transformation is problematic for sites that experience zero flow, because the logarithm of zero is undefined. Gronsdahl et al. (2019) excluded all zero values from their day-of-year analysis, while Perry and Jones (2017) and Segura et al. (2020) set zero values to 0.01 prior to transformation.

Several studies used the number of low-flow days each year as an index. This approach has the advantage that it accommodates the inclusion of zero-flow values in a simple manner, since they are counted as low-flow days. An important consideration is the choice of threshold for identifying low-flow days. Keppeler and Ziemer (1990) used a threshold of 5.66 L s^{-1}, but they did not provide a rationale for this value or why it differed from the threshold used to define a low-flow period. Harr (1980) used a threshold of 0.11 L s^{-1} ha^{-1}, while Harr et al. (1982) used 0.022 L s^{-1} ha^{-1}, both of which were stated to be "arbitrary" (Harr, 1980, p. 5; Harr et al., 1982, p. 640). In both Harr (1980) and Harr et al. (1982), the number of low-flow days reached up to almost 150, or almost five months. This range indicates that the threshold for defining low-flow days was sufficiently high that they extended outside the July–September period, which is often of most interest in relation to aquatic habitat. Gronsdahl et al. (2019) counted the number of days during the July 1–September 30 period with flows less than 10% of long-term mean annual discharge (MAD). This threshold was chosen because it is commonly used to define the point at which further reductions in discharge result in severe degradation of aquatic habitat (Tennant, 1976).

Assessing the significance of treatment effects in paired-catchment studies

One approach to assessing the significance of a treatment effect in a paired-catchment analysis is to use analysis of covariance to test whether the relation between the low-flow indices for the treatment and

Volume 4 No. 01 Moore, Gronsdahl, & McCleary . (2020). Effects of Forest Harvesting on Warm-Season Low Flows in the Pacific Northwest: A Review. http://confluence-jwsm.ca/index.php/jwsm/article/view/35/7 . doi: 10.22230/jwsm.2020v4n1a35

control catchments differs significantly between pre- and post-treatment periods. This approach involves fitting two models to the full data set: 1) a reduced model with the treatment catchment data as the response variable and the control catchment data as a predictor, and 2) a full model that includes, as additional predictors, a factor variable differentiating pre- and post-harvest periods and its interaction with the control stream data. This approach was used, for example, by Gronsdahl et al. (2019), Moore and Scott (2005), Swanson et al. (1986), Troendle et al. (2001), and Winkler et al. (2017), for indices including monthly water yield and number of low-flow days.

An alternative approach is to compute prediction limits and to determine whether post-harvest values fall outside the limits (which would indicate a statistically significant effect for an individual year). Harris (1977) based his evaluation on whether or not the mean observed response for the post-harvest period fell within the prediction limits. However, this approach is not correct; the mean observed response does not need to lie outside the prediction limits for a change to be significant. Harr et al. (1982) and Keppeler and Ziemer (1990) subjectively inferred significance based on the frequency and temporal pattern of post-harvest observations that fell outside the 95% prediction limits. A more rigorous approach would be to use the binomial distribution to compute the probability of finding the observed frequency of points falling outside the prediction limits, or more, over the post-harvest period (Som et al., 2012).

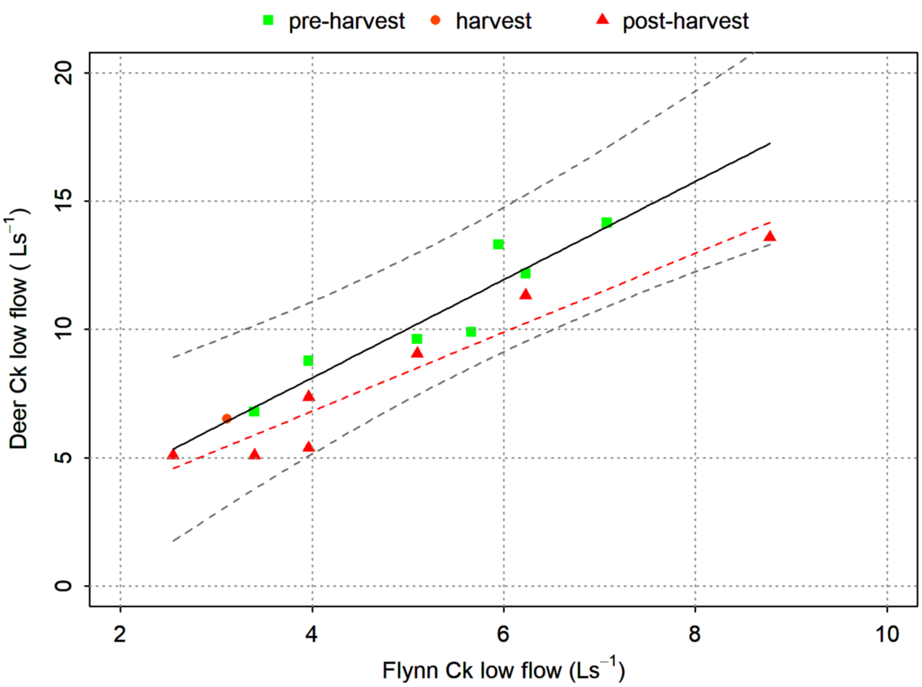

Figure 1. Paired-catchment analysis applied to the minimum August–September daily streamflow for Deer Creek. Flynn Creek was the control catchment. The data were extracted from Table 6 in Harris (1977) and converted from cubic feet per second to litres per second. The solid line is the pre-harvest regression, the dashed grey lines are 95% prediction limits, and the dashed red line is the post-harvest regression, which is significantly different from the pre-harvest regression.

Consider, as an example, the case of Watershed 7 in the H.J. Andrews (HJA) Experimental Forest in the Oregon Cascades, which received a shelterwood cut involving the removal of 60% of the basal area. As shown in Figure 6 of Harr et al. (1982), four of the six post-harvest values for the number of low-flow days plotted below the lower 95% prediction limit for the pre-harvest regression. Under the null hypothesis of no treatment effect, the probability that the number of low-flow days would fall outside the 95% prediction limits in any single year is 0.05. In a sample of six years, the probability of finding four or more years with low flows falling outside the 95% prediction limits is less than 10^{-4}, which is clearly statistically significant at $p < 0.05$.

The prediction-limit method of change detection may not be as powerful as the use of analysis of covariance to compare the pre- and post-harvest regression equations. For example, Figure 1 presents a paired-catchment analysis for Deer Creek in the Alsea Watershed Study in the Oregon Coast Range (Harris, 1977). The p-value for the comparison of the pre- and post-harvest regressions is 0.022, which indicates a significant effect at $p < 0.05$. However, none of the post-harvest observations fell outside the pre-harvest prediction limits, which would be interpreted as no significant effect. Based on this example, it can be seen that the approach used by Harris (1977) may lead to incorrect inferences. A disadvantage of comparing the pre- and post-harvest regressions is that the analysis of covariance is based on the assumption that the variances around the regression lines are equal, which may not be valid. The prediction-limit approach, on the other hand, should be robust to changes in variance.

Results for small catchments

Notes on data sources

Table 1 provides a key for the information in Tables 2 and 3, which provide a summary of studies that have addressed low-flow response to forestry operations in small catchments, all of which were based on a paired-catchment approach. Note that Hinkle Creek South Fork has been included with the small catchments, although its catchment area is slightly higher than 10 km^2.

Table 1. Summary of abbreviations used in Tables 2 and 3

Key for treatment types	
Abbreviation	Explanation
BB-x	Broadcast burned over x% of catchment area
CB	Cutblock
CC-x	Clearcut (% of catchment area)
CPB-x	Cable piled and burned over x% of catchment area
CY-x	Cable yarded over x% of catchment area
PC-x	Patch-cut (% of catchment area)
R-x	Road coverage (% catchment area)
RD-x	Road density (km/km^2)
SC-x/y	Shelterwood-cut (x % of catchment area/y % of basal area)
SR-x	Skid road coverage (% of catchment area)
TPB-x;	Tractor piled and burned over x% of catchment area
TY-x	Tractor yarded over x% of catchment area
VR-x	Variable retention (x% of canopy removed)
Key for forest types	
Abbreviation	Explanation
Al	Alder
As	Aspen
DF	Douglas fir
ES	Engelmann spruce
ESSF	Engelmann spruce-subalpine fir
GF	Grand fir
LP	Lodgepole pine
MC	Mixed conifer
SF	Subalpine fir
SG	Second growth
WH x-y	Western hemlock (age range from x to y in years)
Key for streamfow indices	
Abbreviation	Explanation
AnnualMDF	Annual minimum daily flow
AugSepMDF	Minimum daily flow in August and September
AugMF	August mean flow
DailyMF	Daily mean flows by calendar day
Jul-SepMF	Mean flow for July to September
LF30	30-day low flow
#LFD	Number of low-flow days (i.e., days with flow below a threshold)
LFvolume	Water yield during the low-flow season
SepMF	September mean flow

In relation to the Hinkle Creek results, Surfleet and Skaugset (2013) did not perform a formal significance test. However, the application of ANCOVA using the data reported in their Table 2 revealed a significant effect (p = 0.03) (Figure 2). In the ANCOVA, both harvest periods were lumped together in consideration of the small sample sizes. Only the results for South Fork Hinkle Creek are considered, as the data for the sub-catchments were estimates based on occasional discharge measurements and recession analysis.

Table 2. Summary of studies focussed on small rain-dominated catchments

Study location	Catch-ment	Area (ha)	Forest type	Treatment	Post-harvest period (years)	Response variable	Response	Reference
Oregon Cascades (HJA RF)	HJA1	96	DF,WH 300-500	CC-100, CY-100; BB-100	22	AugMF	Increased for 3 years, switching to decrease	Hicks et al. (1991)
					52	Jul-SepMF DailyMF	Increased for 5-10 years, switching to decrease in most of the post-harvest period	Perry & Jones (2017)
	HJA3	101	DF,WH 300-500	R-2.7 km/km^2; PC-30, CY-30, BB-30	25	AugMF	Increased for ~14 years, then mixed increases and decreases	Hicks et al. (1991)
						DailyMF	Increased for first 10 years, shifting to a decrease, esp. for July-August	Perry & Jones (2017)
	HJA6	13	DF 120-130	CC-100,CY-93, TY-7	6	#LFD	Decreased for entire post-harvest period	Harr et al. (1982)
					38	Jul-SepMF DailyMF	Increased for 22 years, switching to decrease for remainder of period	Perry & Jones (2017)
	HJA7	15.4	DF 120-130	SC-100/60, CY-40,TY-60, BB-100	6	#LFD	Decreased for entire post-harvest period	Harr et al. (1982)
					38	Jul-SepMF DailyMF	Increased for 3 years, switching to decrease at ~year 7 for rest of post-harvest period	Perry & Jones (2017)
	HJA10	10.2	DF,WH 300-500	CC,CY-100	36	Jul-SepMF DailyMF	Increased for ~10 years, switching to decrease for rest of post-harvest period	Perry & Jones (2017)
Oregon Coast Range (AWS)	Needle Branch	70.8	A, DF 120	CC-82, TY-10, BB-82, R-5%	7	AugSepMDF	Initial increase trending down to decrease for last two years (ns)	Harris (1977)
					13 (34 to 46 years post-harvest)	AugSepMDF	Four zero-flow values, not observed in original Alsea Watershed Study	Stednick (2008)
				Second entry, CC-48	5 following first phase of second logging pass	DailyMF for June 1 to September 15	Increased 76% relative to end of first post-harvest period (2006-2009) but still 21% below original pre-harvest levels	Segura et al. (2020)
				Second entry, CC-96	3 following completion of second logging pass	DailyMF for June 1 to September 15	49% higher relative to end of first post-harvest period (2006-2009), but 36% lower than original pre-harvest levels	Segura et al. (2020)

Table 2 (continued)

Study location	Catch-ment	Area (ha)	Forest type	Treatment	Post-harvest period (years)	Response variable	Response	Reference
	Deer Creek	304	A, DF 120	PC-25, BB-8, R-4%	7	AugSepMDF	Decreased for entire post-harvest period	Harris (1977)
					13 (34 to 46 years post-harvest)	AugSepMDF	Figure 9.8 suggests greater variability than pre-harvest	Stednick (2008)
					7	DailyMF for June 1 to September 15	4% increase	Segura et al. (2020)
					12 (40 to 53 years post-harvest)	DailyMF for June 1 to September 15	14% decrease	Segura et al. (2020)
Oregon Cascades (Fox Creek)	FC1	59	DF, WH 300-500	PC-25, CY-25, R, BB-25	9	#LFD	Increased for entire period	Harr (1980)
	FC3	71	DF, WH 300-500	PC-25, CY-19, TY-6	8	#LFD	Increased for entire period	Harr (1980)
Oregon Cascades (Coyote Creek)	CC-1	69.2	DF, MC 100-300	SC-100/50, TY-100, R-1.6%	43 (first 12 and last 12)	DailyMF	Increased in Aug-Sept for first 12 years, near-zero change for last 12	Perry & Jones (2017)
	CC-2	68.4	DF, MC 100-300	PC-30, CY-16, TY-14, CPB-16, TPB-14	43 (first 11 and last 13)	DailyMF	Increased in Aug-Sept for first 11 years but trending toward zero toward end of record	Perry & Jones (2017)
	CC-3	49.8	DF, MC 100-300	CC-100, CY-77, TY-23, CPB-77, TPB-23	41 (first 6 and last 12, no data in between)	Jul-SepMF DailyMF	Increased for first 6 years; decreased for last 12	Perry & Jones (2017)
Carnation Creek, Vancouver Island, British Columbia	Sub-basin H	12	CWH	CC-90, R-6.5%, BB-35	3	#LFD, AnnualMDF	#LFD decreased (ns); AnnualMDF increased	Hetherington (1982)
Caspar Creek, northern California	South Fork	424	RW, DF 85 (SG)	SC-100; TY-100 (in 3 stages)	13	Daily flow on the final day of the low-flow period, #LFD; LFvolume	All indices increased for 5 to 10 years, then decreased	Keppeler & Ziemer (1990)
Hinkle Creek, Oregon Cascades	South Fork	1084	DF WH	CC-26 (in 2 stages)	5 (3 + 2)	AugMF	Increased in all years	Surfleet & Skaugset (2013)

Notes: “ns” indicates response is not statistically significant; HJA RF = H.J. Andrews Research Forest; AWS = Alsea Watershed Study

Table 3. Summary of studies focussed on small snow-dominated catchments

Study location	Catch-ment	Area (ha)	Forest type	Treatment type	Post-harvest period (yrs)	Response	Reference
Wagon Wheel Gap, Colorado	B	81	ES, DF, As	CC-100	7	LF30–no significant change; Baseflow recession flows increased 11%	Van Haveren (1988)
Wyoming	Coon Creek	1673	ES, SF, LP	CC-24 (240 small units)	5	AugMF increased 14% (ns); SepMF increased 20% (ns)	Troendle et al. (2001)
East side Rocky Mountains, Alberta	Cabin Creek	236	ESSF >200; Al; LP	CC-21 (6 units)	8	AugMF increased 4% (ns); SeptMF increased 11% (ns)	Swanson et al. (1986)
UPC WE, British Columbia	241 Creek	490	ESSF	CC-45, R-2.7%	7	AugMF decreased 20% (ns); SepMF decreased 4% (ns)	Winkler et al. (2017)
					10	#LFD increased by 6 days (ns); Jul-SepMF decreased 16%; Persistent decrease in DailyMF from Jun-Aug then increase in DailyMF in Sept	Gronsdahl et al. (2019)
UPC WE, British Columbia	Dennis Creek	373	ESSF	CC-48, R-4%	17	Jul-SepMF decreased by 24% 1-8 yr post-harvest (ns), then decreased 43% 9-17 years post-harvest; #LFD increased 4 days 1-8 years post-harvest (ns) and increased 19 days 9-17 years post harvest; Sporadic and persistent decreases in DailyMF from Jun-Sep	Gronsdahl et al. (2019)
Horse Creek, central Idaho	6 small catch-ments	28.3-147.7	MC, GF	R-1.8% to R-4.3%	1 or 2	No significant changes in AnnualMDF; increases in all catchment-years in flows equalled or exceeded 75% of the time (ns)	King & Tennyson (1984)

A number of the catchments have been used in multiple publications. For example, Harr (1983), Jones and Post (2004), Perry and Jones (2017), and Rothacher (1970), among others, have reported results for watersheds 1 and 3 in the HJ Andrews Experimental Forest. This review focussed primarily on studies that reported monthly water yields for August or September and/or a specific low-flow metric, such as the seven-day low flow or the number of days with flow below a set threshold. In the review below, results for catchments with rain-on-snow regimes are considered together with rain-dominated catchments.

Rain-dominated catchments

Most studies of rain-dominated catchments found statistically significant increases in warm-season low flows within at least the first five years after harvest. For example, Keppeler and Ziemer (1990) found that the discharge on the last day of the low-flow period was increased for the first eight years following harvest but then decreased. Similarly, at Hinkle Creek South Fork, August streamflow increased for all five post-harvest years (Surfleet & Skaugset, 2013). Perry and Jones (2017) also found increased daily flows during summer for eight catchments in Oregon over at least the first three years following harvest.

The direction of responses varied in some cases with the low-flow index used in the analysis. At Needle Branch in the Oregon Coast Range, the minimum August–September daily flow increased for five years post-harvest and then decreased, although the changes were not statistically significant (Harris, 1977); this result was confirmed using ANCOVA. However, one must be careful in the interpretation of non-significant results, as a lack of significance could result from high variance and the relatively small sample size (six years pre-harvest and seven years post-harvest). A re-analysis of the initial post-logging period for Needle Branch showed that summer (June 1–September 15) mean daily flows were, on average, 3% below pre-harvest levels, but no assessment of statistical significance was provided (Segura et al., 2020).

At Deer Creek, which had a much smaller area harvested compared to Needle Branch, Segura et al. (2020) found that summer daily flows averaged 4% above pre-harvest levels for the first seven years following harvest, but they did not provide an assessment of statistical significance. In contrast, the minimum August–September daily flow at Deer Creek was significantly lower than predicted for the first seven post-harvest years (Harris, 1977). Although Harris (1977) indicated that the response was not significant, based on the post-harvest mean response plotting within the 95% prediction limits, a re-analysis of the data using ANCOVA indicates a significant effect with $p = 0.02$ (Figure 2). This response could reflect the relatively low extent of harvesting (25% of the catchment area) and the fact that riparian buffers were retained along the streams (both of which would reduce the magnitude of post-harvest flow increases relative to more extensive clearcutting with no buffer), coupled with the effects of the road network, which covered 4% of the catchment area. As mentioned in the section titled "roads," the effect of roads would likely be to reduce baseflow contributions to streamflow.

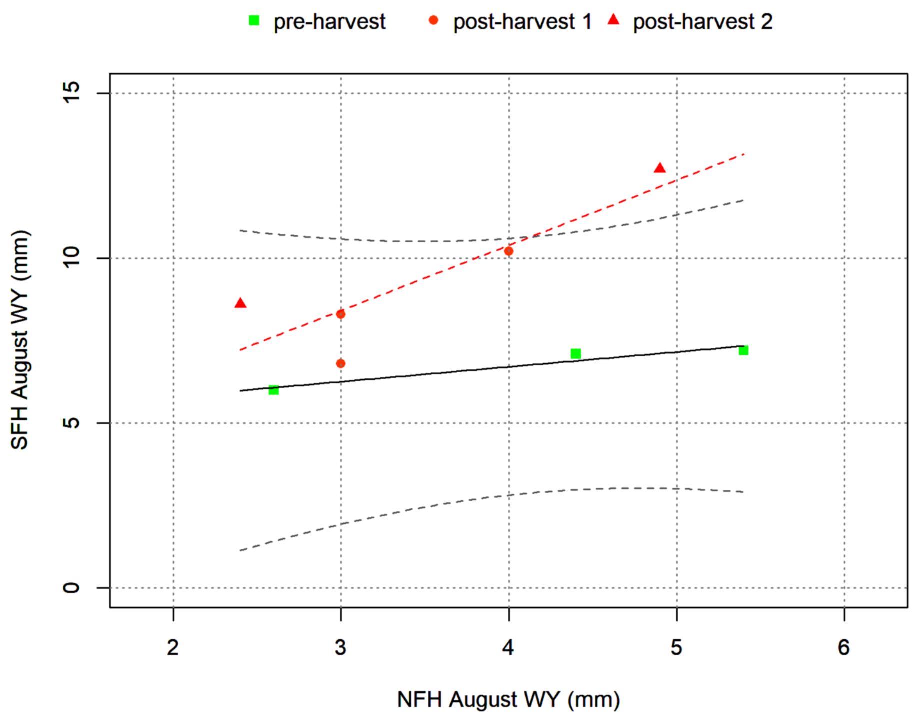

Figure 2. Paired-catchment analysis for August water yield at South Fork Hinkle Creek (SFH). North Fork Hinkle Creek (NFH) is the control catchment. Data are from Table 2 in Surfleet and Skaugset (2013). The solid line is the pre-harvest regression, the dashed grey lines are 95% prediction limits, and the dashed red line is the post-harvest regression, which is significantly different from the pre-harvest regression.

An interesting case is the Fox Creek catchments, where significant increases in the number of low-flow days were found in the first six years following harvest (Harr, 1980). This response was attributed to a loss of water inputs by fog drip, corroborated by measurements of throughfall and rainfall (Harr, 1982). Ingwersen (1985) presented evidence that, as forest regenerated, fog drip recovered and low flows appeared to increase. Ingwersen (1985) found that the effects of reduced fog drip were only observable for June and July. As mentioned in the section titled "results for rain-dominated catchments," the low-flow index used by Harr (1980) did not isolate the July–September period. Therefore, further analysis would be required to determine the response for flows in August and September, when the lowest flows typically occur.

A complication in analyzing post-harvest response to forestry is that the treatment effect varies from year to year as a function of hydroclimatic conditions. Keppeler and Ziemer (1990) reported both significant increases and decreases in the number of low-flow days in the first five years post-harvest at Caspar Creek in northern California. Through multiple regression analysis, they found that post-harvest increases in low flows were negatively related to the frequency of days with rain during the low-flow period. They suggested that, under conditions with light rain or fog, transpiration from the forest in the control catchment would be reduced, leading to less difference between the treatment and control catchments. An additional factor is water input by fog drip, which is common in northern California, and would be reduced following harvest, as was demonstrated for Fox Creek.

The post-harvest increases in low flows for rain-dominated catchments varied substantially in their duration, from five or fewer years in some cases to two decades or more in others (see Table 2). Perry and Jones (2017) noted that the magnitude of flow increases was smaller in thinned stands than for clearcuts, and that the increases disappeared more rapidly. The variability in magnitude and duration of response also may be linked to differences in post-harvest succession. For example, alders became established in the relatively wide valley floor at H.J. Andrews Watershed 1, leading to a rapid increase in transpiration loss, whereas conifers regenerated along the stream at Watershed 3, allowing the post-harvest increases to persist (Hicks et al., 1991). Unfortunately, detailed information on post-harvest succession is not available for many of the study catchments.

A number of paired-catchment studies in Oregon now have sufficiently long records that facilitate the examination of responses over multiple decades. For example, Perry and Jones (2017) found that, of eight catchments with records extending more than 36 years following harvest, six exhibited flow decreases in at least the last few years of record. Segura et al. (2020) similarly found that summer daily flows were reduced relative to pre-harvest conditions for both Needle Branch and Deer Creek, where regenerating forest stands between 40 and 53 years of age were associated with warm-season low flows averaging 50% below pre-harvest levels for the 82% clearcut Needle Branch catchment and 14% below average for the 25% patch-cut Deer Creek catchment.

Perry and Jones (2017) and Segura et al. (2020) presented low-flow responses as multi-year averages. However, it appears that a lingering effect of harvest may be to increase the sensitivity of the catchment to hydroclimatic conditions, generating greater interannual variability. For the period 34 to 46 years following harvest for Deer Creek, Figure 9.8 in Stednick (2008) shows four points plotting above the upper 95% prediction limit and four below the lower prediction limit, for a total of eight exceedances over 13 years. Using the binomial distribution, the p-value for this result is less than 10^{-7}, i.e., highly significant. Stednick (2008) noted that in the period from 34 to 46 years after harvest, Needle Branch experienced four years with zero-flow days, which had not been observed in the original study period.

Segura et al. (2020) also reported the effects of logging second-growth forest in the Needle Branch catchment, when the plantation forest was 40- 58-years old. The effect of the logging activity was to increase warm-season daily flows by 50% or more relative to conditions in the four years just prior to the second pass of logging. However, flows were still reduced relative to conditions prior to the original harvest, when the catchment was dominantly covered by mature or old-growth forest.

To the authors' knowledge, only one catchment-scale study—Watershed 3 in the HJ Andrews Experimental Forest in the Oregon Cascades—maintained a road-only treatment for a sufficient length of time to allow the effect on streamflow to be estimated. After a six-year pre-treatment calibration, a road was constructed, followed by a four-year gap until harvesting. There was no obvious effect on annual water yield, but the response of a low-flow-related metric was not analyzed (Rothacher, 1970).

Snow-dominated catchments

For the catchments with snow-dominated regimes, statistically significant results were reported in studies by Gronsdahl et al. (2019) and van Haveren (1988), which involved the highest levels of harvesting (100% for Wagon Wheel Gap and 48% for Dennis Creek). The directions of change were opposite, with a significant increase in baseflow for Wagon Wheel Gap and a reduction in low-flow indices at Dennis Creek. Note, however, that there was no significant change in the 30-day low flow at Wagon Wheel Gap. A key point is that Dennis Creek was the only catchment with data spanning more than a decade following the completion of harvesting, and the significant responses occurred toward the end of the post-harvest period. Winkler et al. (2017) reported non-significant declines in August and September streamflow for 241 Creek, which, like Dennis Creek, is part of the Upper Penticton Creek Watershed Experiment (UPC WE).

In the other two studies that involved forest harvesting in snow-dominated catchments (Coon Creek and Cabin Creek), increases in August and September water yield were observed, albeit not statistically significant. One challenge with interpreting the non-significant results in these cases is that both catchments experienced relatively low levels of harvest (21% and 24%, respectively). Detecting the effect of these low-intensity treatments is compromised by the relatively low sample size and relatively high variance about the pre-harvest regression lines. Indeed, Swanson et al. (1986) calculated that over 25 years of post-harvest data would be required to detect an increase of 10% in August streamflow at a 0.1 significance level. A further complication is that, even if a longer post-harvest period were available, it is likely that the treatment would have diminished or even changed direction over two or more decades in association with forest regrowth.

King and Tennyson (1984) evaluated the effect of road construction (no harvest) on streamflow in Idaho over the first one (for six catchments) or two (for three catchments) years using the prediction-limit approach. There were no significant effects on the annual minimum daily flow, which typically occurs in September or October (King, 1989). There were also no significant changes in Q_{75}, which King and Tennyson (1984) defined as the flow that is exceeded 75% of the time. (Note that this definition is potentially confusing, because Q_{75} is more commonly used to indicate a flow that is exceeded 25% of the time). This flow typically represents conditions in late summer in Idaho. Despite the lack of statistically significant changes, increases were reported for both years in all catchments. There were only four years of pre-harvest data, so prediction limits would have been wide, leading to low statistical power, and the consistency of response suggests that the increase in Q_{75} (which was up to 82% of the predicted flow) could be real even if not statistically significant. If this effect were real, it could reflect an increase in stormflow generation during summer rainstorms following road construction. It would be useful to follow up with a more detailed analysis to quantify whether the rainfall-runoff relation changed following road construction.

Results for medium to large catchments

Bowling et al. (2000) analyzed streamflow time series for 23 catchments in the Washington Cascades with drainage areas from 14 to 1600 km^2. Although not explicitly stated, these catchments likely have rain-dominated or hybrid regimes. Bowling et al. (2000) computed difference time series for each catchment using the catchments with the lowest rates of vegetation change as reference series. The results varied with the time period considered. For 1960–1996, 11 out of 18 catchments exhibited significant increasing trends in the annual minimum flow, while five exhibited significant negative trends. However, considering the longer period 1945–1996, five of eight catchments with data exhibited negative trends in annual minimum flow. For 1930–1996, all three catchments with data for that period exhibited negative trends, but only one was statistically significant.

Moore and Scott (2005) updated an earlier analysis by Cheng (1989) of the response of Camp Creek to salvage harvesting following a mountain pine beetle infestation, using data up to and including the year 2000. Camp Creek drains the uplands west of Okanagan Lake and has a catchment area of 33.9 km^2 at the location of a Water Survey of Canada stream gauge. Streamflow records for an adjacent catchment, Greata Creek, were used as a reference time series for a paired-catchment analysis. Water Survey of Canada metadata indicate that Greata Creek has unregulated flow at the location of the stream gauge. Furthermore, an analysis of forest harvesting and mortality caused by mountain pine beetle indicate that the equivalent clearcut area (ECA) remained below 3% until about 2005 (Qiang Li, 2018, personal communication), supporting the validity of using Greata Creek as a reference for the period analyzed by Moore and Scott (2005). Neither mean monthly flow nor seven-day minimum flow in August or September showed a significant effect according to ANCOVA. However, the differences between observed and predicted mean monthly flows in August were dominantly positive for the first 13 years following harvest, shifting to negative for the next five years (1996–2000). An updated analysis following the approach of Moore and Scott (2005) indicates that the negative differences persisted through 2015, suggesting that low flows had become chronically more extreme (Figure 3).

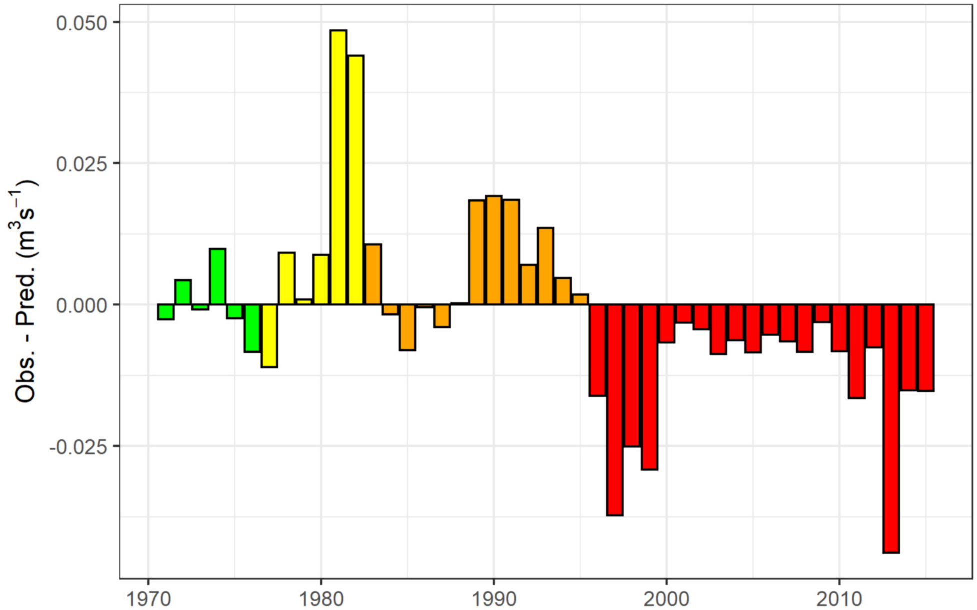

Figure 3. Differences between observed August streamflow and streamflow predicted from the pre-harvest regression for Camp Creek, B.C. Green bars indicate the pre-harvest period and yellow bars indicate the period in which harvesting occurred. The orange and red bars indicate the post-harvest periods before and after the breaching of the dam on Glen Lake. Positive (negative) values indicate an increase (decrease) in August streamflow relative to pre-harvest conditions.

The apparent shift to more extreme low flows at Camp Creek in August must be interpreted cautiously. Contrary to the reported status of Greata Creek as unregulated at the Water Survey of Canada gauge, further investigation revealed that a dam had been constructed at Glen Lake, in the upper portion of Greata Creek's catchment, to support a water license granted in 1913 to the District of Peachland (Golder Associates, 2010). The dam was breached in 1996 due to increased costs to maintain dam safety and the fact that the water storage was not being used. A new dam was constructed in 2010 (Layh, 2010). It is entirely plausible that the shift from increased to decreased August streamflow in 1996 is spurious. For example, suppose August streamflow at Camp Creek were substantially unchanged compared to pre-harvest conditions. In this case, an increase in flows at Greata Creek, relative to conditions in the pre-harvest period (i.e., prior to dam breaching), would result in the paired-catchment analysis indicating a spurious decrease in flows at Camp Creek relative to predictions from the pre-harvest regression.

Zhang and Wei (2012) applied trend analysis and correlation analysis between the seven-day low flow and ECA for Baker Creek, a 1570 km^2 snow-dominated catchment in the British Columbia Interior. The authors did not comment on the timing of the seven-day low flow, but, considering the hydroclimatic context, it likely occurs in winter and is thus beyond the scope of this review.

Zhang and Wei (2014) focussed on the snow-dominated Bowron and Willow catchments in central British Columbia, which have respective catchment areas of 3420 and 2860 km^2. Similar to Zhang and Wei (2012), they computed ECA as an index of the cumulative effect of forest disturbance through time. Relevant to this review, the seven-day low flow in the warm season (July–October) was computed. Unfortunately, both gauging stations were relocated in 1976 or 1977. Even though the flow records were adjusted based on the change in drainage area, such an inhomogeneity is a cause for concern when attempting to detect the effects of changes in LULC (Troendle & Stednick, 1999). Based on intervention analysis and cross-correlation analysis with the ECA index, there was no significant impact of forest disturbance on the seven-day warm-season low flow.

Linking low flows to aquatic habitat

Linkages between low flows and fish populations are complicated by the fact that the productivity of fish habitat can be limited by factors other than streamflow. For example, the availability of suitable hydraulic habitat depends not only on streamflow but also on channel morphology and substrate characteristics. Other potentially limiting factors include stream temperature and prey availability (Leach et al., 2012). Water-quality indices, including alkalinity, are also an important determinant of fish biomass (Ptolemy, 1993). Further complications arise because forestry activities can influence habitat not only by changing the low-flow regime but also by modifying other factors that control habitat suitability, for example by increasing peak flows, which in turn can potentially cause bed scour and modify channel morphology (Hassan et al., 2005a; Moore & Wondzell, 2005). Forestry also influences in-stream wood dynamics (Hassan et al., 2005b), fine sediment dynamics (Gomi et al., 2005), stream temperature (Moore et al., 2005a), and nutrient and organic-matter dynamics (Rosenfeld, 2017). Complicating matters is that some of these factors may be exacerbated by declines in streamflow. For example, Hicks et al. (1991) reported that the decay of deciduous leaves falling into a stream resulted in oxygen depletion, the extent of which was inversely related to streamflow, while Moore et al. (2005b) found that daily maximum stream temperature within a clearcut varied inversely with streamflow.

Considering the range of influences just described, changes in summer low flows may not limit aquatic productivity in systems in which other factors, such as summer temperatures or nutrients, provide the first-order limit. Forestry-related effects on low flows may also not be as critical in other stream reaches that are fed by contributions from lakes, aquifers, and other storage features that sustain summer low flows above thresholds associated with productive habitat, such as 30% of long-term mean annual discharge (Tennant, 1976).

Linkages between low flows and aquatic habitat are likely to be most significant in catchments with summer flows that drop toward or below established thresholds. For example, Tennant (1976) suggested that a threshold of 10% of long-term mean annual discharge flow corresponded to a higher likelihood of habitat degradations, including harmful changes to stream temperatures. While these flow thresholds may provide a general guide for identifying potential susceptibility to changes in low flows, optimal habitat availability in larger streams generally occurs at a lower percentage of mean annual discharge (MAD) than it does in small streams (Hatfield & Bruce, 2000). Even within the subset of flow-sensitive catchments, linking forestry effects on low flows to impacts on aquatic habitat can be challenging because channel morphology and thus hydraulic conditions can be modified by hillslope processes and riparian area condition, both of which can be influenced by forestry activities (Reid et al., 2020).

Within the subset of flow-sensitive catchments that support forestry, the level of harvest may not necessarily reach the levels associated with declines in summer streamflow in experimental catchments. However, due to climate change and related forest disturbances, including fire and insect infestations, the extent of forest stands associated with significant summer streamflow deficits may be expanding. For example, in the Southern Interior of British Columbia, widespread lodgepole pine mortality due to fire and insect infestations has been occurring since the 1990s in catchments that support known flow-sensitive fish populations. As a consequence of this mortality and subsequent salvage operations, the proportion of watershed assessment units within the Kamloops Timber Supply Area with a moderate, high or very high likelihood of harmful hydro-geomorphic events (e.g., floods, bank erosion, channel instability, debris floods and debris flows) increased from approximately 21% to 40% of all units between 2003 and 2014 (Lewis, 2016). While these hazard indices are focussed on the effects of changes to peak flows, they could potentially be modified to identify which catchments are likely to experience disturbance-related decreases in low flows, particularly as forests regenerate in the disturbed areas.

Penaluna et al. (2015) explored the effects of forestry and climatic change on cutthroat trout populations in northwest Oregon through application of a simulation model that tracked individual trout through their life cycles. To account for the range of effects of forestry on aquatic habitat, boundary conditions such as streamflow and water temperature were adjusted based on a synthesis of published studies in the region. For streamflow, the scenarios were based on an assumption that flows from July to September were increased by 45% above baseline following harvest for a period of five years, followed by a decline back to baseline over the next 15 years, with a continuation of baseline flows until the next harvest. This scenario is not supported by the results of Perry and Jones (2017) and Segura et al. (2020), which suggest a more complex trajectory for summer low flows, as synthesized by Coble et al. (2020) into a three-phase sequence comprising an initial period of increased low flows, an intermediate period of declining flows, and finally a phase with reduced low flows relative to pre-harvest conditions.

Gronsdahl et al. (2019) used the results of their day-of-year analysis, in conjunction with relations between weighted usable area (WUA) and streamflow for rainbow trout, to quantify the magnitude and timing of post-harvest changes in WUA. The WUA-streamflow relations were generated based on field surveys and hydraulic modelling. Beginning about 15 years after the onset of harvesting in one of the catchments, modelled fish habitat availability decreased by about 20%–50% during the summer low-flow period.

Segura et al. (2020) reported that fish biomass had recovered to pre-harvest levels by the end of the first post-treatment period at Needle Branch, despite reduced summer flows. This result suggests that the initial post-harvest decline was the result of other factors, such as the marked increases in summer water temperature and suspended sediment concentrations, both of which recovered toward pre-harvest levels prior to the second logging pass (Beschta, 1978; Harris, 1977). Following the second pass of logging, the late-summer total biomass of coastal cutthroat trout increased (Bateman et al., 2018). The second logging pass generally conformed to contemporary harvesting practices, such as the retention of riparian buffers, which reduced the impacts on stream temperature and suspended sediment concentrations (Bladon et al., 2018; Hatten et al., 2018). Therefore, the response of fish populations to the second logging pass is consistent with the observed increase in summer flow and the likely increase in physical habitat availability.

Discussion and Recommendations for Further Research

Synthesis of previous studies

Results for the small rain- and rain-on-snow-dominated catchments in Oregon and California are generally consistent with the inferences based on stand-level research, with an initial increase in warm-season low flows that was followed, for a number of catchments, by a longer-term decline that can apparently last for several decades. The duration of the increased flows was highly variable, from fewer than five years to more than two decades. The two exceptions to this dominant pattern are Fox Creek, where the reduction of fog drip following harvest resulted in more frequent low-flow days for the first five post-harvest years, and Deer Creek, where the post-harvest decrease in August–September minimum daily flow could reflect the relatively low level of harvest, the retention of riparian buffers, and a reduction in baseflow contributions due to the road network.

The variability in responses is due, in part, to differences in the rate and pattern of post-harvest succession, as well as to differences in the harvest treatments (such as the percent of the catchment that was harvested, whether harvesting was clearcutting or partial-harvest, whether hauling was ground-based or by high lead cable system, and whether slash was broadcast-burned). In addition, the spatial pattern of harvest may be a factor. For example, transpiration from riparian forest can have a greater effect on streamflow than transpiration from upslope stands as was shown, for example, for a rain-dominated catchment in South Africa (Scott, 1999). Therefore, catchments with riparian buffers may experience less of an initial post-harvest increase than those without.

Post-harvest responses for the five small snow-dominated catchments were mixed, with two showing non-significant increases in flow in August and September, two showing decreases (one significant), and one with a significant increase in baseflow. Data extended beyond the first decade following the completion of harvesting operations for only one catchment. For that case, significant reductions in late-summer flow were found toward the end of the study period.

For both rain- and snow-dominated catchments, it appears that the pattern of post-harvest low-flow response over decadal time periods can be inconsistent with current approaches to modelling hydrologic recovery, which are typically based on the assumption of a monotonic and asymptotic trend toward 100% recovery, often parameterized as a function of tree height (e.g., Hudson, 2000; Lewis & Huggard, 2010; Winkler & Boon, 2015). Therefore, the use of indices such as ECA, as currently calculated, may not be effective covariates for representing the effect of forest cover on low-flow variations through time. Instead, the hydrological response to low flows may be better represented by an increase in flow magnitude in the short term, a reduction in flow during the medium term as juvenile forests regenerate, and finally a trend back toward pre-harvest conditions as the forests mature. This trajectory has been observed for annual water yield in Australia and modelled using a nonlinear function that is locally termed the "Kuczera curve" (Kuczera, 1987; Vertessy et al., 2001; Watson et al., 1999).

One challenge in synthesizing the results of previous studies in terms of general post-harvest trajectories is that, in a number of studies, the treatment occurred over several years. For example, at 241 Creek, harvesting operations occurred from 1993 to 2007 (Gronsdahl et al., 2019). Considering how rapidly harvesting effects can change over periods of five to ten years, some cutblocks could be generating more streamflow than for pre-harvest conditions, while others could either be "recovered" or even producing less streamflow.

There are two limitations to the transferability of information from paired-catchment studies. One is that the hydrologic effects of forestry are strongly influenced by site-specific factors such as topography, soils, and geology, as well as details of the forestry treatments, such as the locations and size of cutblocks and the locations and design of road networks. As a consequence, it is difficult to extrapolate the results from paired-catchment studies to other sites or treatments. The second is that paired-catchment studies have often treated the catchments as block boxes; while they allow the quantification of treatment effects, on their own they do not provide insights into the governing processes, which is required to guide the transfer of information from experimental catchments to a broader population of catchments. In other words, paired-catchments permit the detection and quantification of changes, but not necessarily attribution.

Future research directions

A key challenge in synthesizing the results of the reviewed studies is the lack of a common low-flow metric. It would be a worthwhile project to revisit all of these studies to compute multiple relevant indices, including the seven-day warm-season low flow, the 30-day minimum flow from July to September (which is used in British Columbia for fish-habitat assessment), and the number of low-flow days using a consistent threshold, such as 10% of long-term mean annual discharge. In addition, it would be valuable to perform these analyses for cases such as Fool Creek and Deadhorse Creek in Colorado (Troendle & King, 1985, 1987), where previous analyses focussed on peak flows and annual water yield, but not low flows.

Most of the results for small catchments were based on clearcutting, either as a single block or as a number of patches. However, thinning and other partial-cut silvicultural treatments are increasingly common. Stand-level results from Bladon et al. (2006) suggest that thinning should produce a similar pattern to clearcutting, simply reduced in the initial magnitude of increase. Results from HJA Watershed 7 and Coyote Creek 1, which both received thinning treatments (Perry & Jones, 2017), support this inference. Further process-based work should particularly focus on the effects of thinning and post-thinning stand development on the water balance, especially for snow-dominated catchments, where this treatment appears to have been less well studied than in rain-dominated catchments. In addition, considering the increasing retention of forested riparian buffer strips in many jurisdictions, further research should focus on the effects of transpiration from riparian buffers on low flows.

The results for the medium-to-large catchments in western Washington considered by Bowling et al. (2000) are intriguing and deserve more detailed attention. In particular, further analysis should focus on relating the variations in low-flow severity to forest-cover conditions through time and across catchments.

This review has identified significant knowledge gaps that highlight inadequacies in the analytical tools that are currently used in watershed management, particularly the calculation of indices such as ECA. While paired-catchment studies can potentially provide direct evidence for the effects of forestry and post-harvest forest stand development on streamflow measures, new studies cannot provide timely information on the long-term impacts of forestry practices, particularly over multiple decades. The maintenance of all existing experimental catchments over the long term is recommended, so they can provide ongoing information on long-term impacts.

To maximize the information that can be gleaned from paired-catchment studies, research should be initiated within existing experimental catchments on key ecohydrological processes, such as interception loss, transpiration rates, snow accumulation and melt, soil moisture dynamics, and runoff generation. Such studies should focus on identifying the spatial and temporal origins of streamflow (e.g., snowmelt versus summer rainfall) and how these vary through space and time. These field studies would not only provide insights into the underlying processes and support the attribution of post-harvest changes in streamflow, but they would also provide critical data for the internal validation of process-based models that can explicitly account for site variations in soils, topography, climate, forest type, and forestry treatment. Such models could potentially be used as management tools to supersede the use of simple indices such as equivalent clearcut area.

Finally, future research should focus on the implications of streamflow changes for fish habitat and the development of tools to support watershed management. Numerous factors are generating an urgency to advance these research priorities, including the impacts of mountain pine beetle and associated salvage harvesting (e.g., Lewis, 2016); the climatic shifts to earlier freshets and prolonged periods of low flows (Pike et al., 2010); increased emphasis on optimizing fiber production in managed forests by utilizing shorter rotations that tend to result in higher evapotranspiration rates, at least in coastal forests (e.g., Segura et al., 2020); and the recent declines of numerous fish stocks that are sensitive to summer low flows (e.g., DFO, 2016). Research priorities include 1) developing methods to classify streams by the primary factors that limit their fish productivity, for example by combining streamflow thresholds (e.g., Tennant, 1976), water temperature thresholds (e.g., Parkinson et al., 2016) and other relevant indices; 2) expanding the spatial scaling capabilities of process-based hydrological models so they can be used to determine how forestry-related changes to summer streamflow will worsen or attenuate at downstream points of interest; and 3) developing silvicultural treatments to promote the recovery of summer streamflow that takes into account the variability in stand dynamics over full rotation cycles. Such silvicultural treatments would also benefit utility managers responsible to agricultural and public water users, who have expressed similar needs (Clark, 2018).

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