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RESEARCH ARTICLE

Forestry impacts on stream flows and temperatures: A quantitative synthesis of paired catchment studies across the Pacific salmon range

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Abstract

1. Forestry is pervasive across temperate North America and may influence aquatic environmental conditions such as flows and temperatures, as well as important species such as Pacific salmon (*Oncorhynchus* spp.). While there have been many large-scale forestry experiments using paired catchment designs, these studies have yet to be quantitatively synthesized. Thus, it remains unclear whether forestry impacts are consistent, context-dependent or unpredictable.
2. This study aims to quantitatively synthesize forestry impacts on streamflow and temperature, through a systematic review and synthesis of paired catchment studies across the range of Pacific salmon. Specifically, we investigated whether generalizable relationships exist between forestry intensity (percent watershed harvested) and impacts to streamflow and temperature. We also examined whether watershed features (climate, hydrology and lithology) and harvest method mediated forestry impacts.
3. We extracted information from 35 unique paired-catchments from California to Alaska. Forestry had strong impacts on peak and low flows and maximum summer water temperatures, but responses were quite variable. Across all catchments, forestry elevated peak flows ~20% ($n=31$ catchments), reduced low flows ~25% ($n=13$ catchments) and increased maximum summer temperatures ~15% ($n=35$ catchments) on average. However, these impacts were variable and were not predictable based on forestry intensity, thus broader stressor–response relationships were not supported.
4. Forestry impacts on peak flows and maximum summer temperatures varied spatially. Peak flow impacts increased with northward latitude and temperature impacts decreased with eastward longitude. However, the magnitude of impacts were unrelated to other watershed attributes, which included climate (precipitation and aridity), rain versus snow hydrology, elevation and bedrock lithology.

Harvest method and riparian buffer presence also had no detected effects on forestry impacts across studies and statistical models explained a low proportion of variation overall.

5. Collectively, our results indicate that forestry can have substantial impacts on key environmental conditions; however, the magnitude of impact was variable and could not be clearly linked to easily measured watershed characteristics. This implies that forestry impacts may not be broadly predictable. Probabilistic risk models based on distributions of potential impacts may therefore be more useful for watershed management in data-poor situations.

KEYWORDS
forestry, hydrology, salmon, watersheds

1 | INTRODUCTION

Watersheds supporting critical social–ecological systems are rapidly changing due to land use activities and climate warming (Vörösmarty et al., 2010). In the face of these changes, there is a need for tools and heuristics to understand and predict how valued ecosystem components respond to activities and associated stressors, including their rates of change and possible thresholds where they become adversely affected (Adams, 2003; Groffman et al., 2006; Rosenfeld et al., 2022). While mechanistic modelling approaches that incorporate hydrological processes are widely used, (e.g. MIKE SHE; Im et al., 2009), empirically based stressor–response relationships can be more tractable in many situations (Jarvis et al., 2024; Pirotta et al., 2022). Stressor–response relationships are often generated from data syntheses that aggregate responses across numerous case studies (e.g. Courtice et al., 2022), with the goal of being generalizable across systems. Yet, watersheds are diverse and complex systems that may differ in their sensitivity and responses to stressors (Dey et al., 2024; McCluney et al., 2014). Thus, it may be possible to identify watershed attributes that influence their sensitivity as part of these synthesis efforts. Stressor–response relationships and sensitivity heuristics can together be powerful management tools, yet they are not well developed for many types of activities.

Forestry has a particularly expansive footprint across much of western North America, as well as globally, and can dramatically alter watersheds and associated hydrological, geomorphic and ecological processes (Moore & Wondzell, 2005; Richardson & Béraud, 2014). Typically, forestry activities such as harvesting and road building can induce immediate increases in peak flows and sometimes base flows due to reduced water retention (Moore & Wondzell, 2005). Then as stands regenerate, increased evapotranspiration can reduce base flows, often decades after harvest (Coble et al., 2020). Forestry also often elevates water temperatures, especially during summer, primarily due to reduced canopy cover and increased solar radiation along, with additional influence from other thermal processes (e.g. warming associated with baseflow reduction; reviewed in Moore et al., 2005). These impacts to flow and temperature pose a risk to aquatic ecosystems and to the viability of culturally and economically important taxa such as Pacific salmon (*Oncorhynchus* spp.). For example, forestry-linked increases in peak flows and decreases in low flows can elevate mortality risk for eggs and diminish habitat quality for juveniles (Gronsdahl et al., 2019; Solazzi et al., 2000). Similarly, forestry-linked increases in water temperature can elevate the risk of thermal stress and mortality (Groom et al., 2017; Pollock et al., 2009). While the realized impacts of these stressors on salmon populations vary, they can be significant. For instance, in Southwest British Columbia, forestry activities have been linked to substantial reductions in survival and productivity for multiple salmon species (Tschaplinski & Pike, 2017; Wilson et al., 2022).

Despite significant progress towards a solid process-based understanding of forestry impacts, there is high uncertainty in the magnitude of forestry impact to salmon watersheds, and generalizable stressor–response relationships remain elusive. This is a critical gap given the prevalence of historical and ongoing forestry across the Pacific salmon range, and the imperilled state of many salmon populations (Gustafson et al., 2007). Improving this situation requires addressing two linked challenges. First, evidence of forestry impacts comes primarily from detailed case studies, with limited attempts at synthesis (but see Grant et al., 2008). Thus, there is a need to bring together information towards a more holistic perspective on impacts across diverse systems. Second, case studies have revealed that impacts are highly context dependent, varying in magnitude and even direction across systems (Brown et al., 2005; Moore & Wondzell, 2005). Therefore, it appears that watersheds vary in their sensitivity to forestry, likely due to numerous watershed attributes that mediate impacts, including physical characteristics such as slope and aspect, bioclimatic conditions such as precipitation and rain versus snow-driven hydrology (Bateman et al., 2016; Moore et al., 2005), and the diversity of forestry practices themselves; for example, clear-cutting versus thinning (Groom, Dent, Madsen, & Fleuret, 2011). These diverse responses further highlight the need for synthesis given that quantifying the relative influence of these modifying factors is challenging for individual case studies with limited replication.

Here, we quantitatively synthesized forestry impacts on water flows and temperatures from over 75 years of experimental paired-catchment studies across much of the range of Pacific salmon in North America. Specifically, we investigated whether there was a generalizable stressor–response relationship between forestry intensity (the percent of a watershed logged) and impacts to streamflow and temperature. We also examined whether harvest method and watershed features (climate, hydrology, lithology and location) mediated the impacts of forestry across systems. A broader goal of this work was to inform risk assessment and general management of forestry activities in salmon watersheds. As such, we focused on hydrologic metrics with established linkages to Pacific salmon performance and viability, including peak flows, summer low flows and maximum summer temperature (Ward et al., 2015; Warkentin et al., 2022; Zillig et al., 2021). This focus differs from other recent work emphasizing hydrological processes and their responses to forest disturbance (Buma & Livneh, 2017; Goeking & Tarboton, 2022). Thus, our effort is also an attempt to strengthen the bridge between the disciplines of hydrology and fish habitat science.

2 | MATERIALS AND METHODS

2.1 | Literature search

We conducted a literature search to identify studies examining hydrological responses to forestry within the historical range of Pacific salmon. There is a large body of work in this area, with a range of study designs. We focused our analysis on paired catchment studies, which estimate forestry impacts by comparing catchments with specific harvest 'treatments' to adjacent unharvested 'control' catchments. The strength of this design is that hydrological changes from forestry can be disentangled from other influences such as climate variability, which can otherwise obscure inferences (Neary, 2016). Constraining our analysis to paired catchment studies also allowed a more standardized interpretation of results, that is, the response of a treatment catchment relative to a control (see Section 2.2 below).

We conducted literature searches on Web of Science and Google Scholar through the open source software Publish or Perish (https://harzing.com/resources/publish-or-perish). We used the following search strings for flows and temperature:

('forest harvest*' OR forestry OR logging OR clearcut* OR 'forest thinning' OR 'forest management') AND (streamflow$ OR 'low flow$' OR 'peak flow$' OR hydrolog*)

('forest harvest*' OR forestry OR logging OR clearcut* OR 'forest thinning' OR 'forest management') AND (stream temperature$ OR 'water temperature$')

We initially screened abstracts to identify articles clearly out of the study scope (e.g. studies outside the geographical range), then examined the full text to apply the following criteria: (1) the study must have been conducted within the historical range of Pacific salmon in North America; (2) The study must focus on forestry, which may include various forms of harvest (e.g. clear-cutting, thinning) and road building. However, we did not consider studies where forestry was combined with wildfire or other forms of vegetation removal. (3) The study design must compare a treatment to a reference or control; and (4) the study was conducted at the catchment scale, which excluded a number of studies on stream temperature that compared upstream to downstream treatments and control reaches within catchments (Groom, Dent, & Madsen, 2011). We imposed one additional criteria for studies on low flow responses, which have a strong temporal component such that impacts are not expected until ~10–15 years post-harvest as vegetation regenerates (Coble et al., 2020; Perry & Jones, 2017). We therefore only selected studies on low flows where impacts could be clearly attributed to greater than 10 years post-harvest.

After the initial screening, we back- and forward-searched through the references to identify additional studies missed in the initial literature search. We also cross-checked our searches against several comprehensive reviews on forestry impacts (Coble et al., 2020; Grant et al., 2008; Moore et al., 2005; Moore & Wondzell, 2005) and conducted targeted searches of government databases to ensure we obtained all relevant publications. In total, we scanned 3558 publications and identified 14 studies that met criteria for peak flows, 6 for low flows and 13 for maximum summer temperature (hereafter 'temperature'). From these studies, there were 35 unique paired catchments (treatment relative to control) for temperature, 31 for peak flow, and 13 for low flow (Table S1). Given the limited sample size for low flow, we did not pursue formal statistical analysis, but still report overall patterns. Catchments encompassed a range of bioclimatic conditions from Alaska to California (Figure 1; Figures S1 and S2), with the majority located in coastal Oregon.

2.2 | Data extraction

Throughout all analyses, we consider an individual catchment as the unit of interest and the response as the change in a given quantity (i.e. flow or temperature) relative to the control catchment in that study. Insufficient information was reported for any data re-analysis, so we used the reported response magnitudes from the original studies. The response metrics themselves were also reported in a number of ways and could not be standardized to a specific unit given the information provided. Instead, we examined responses on a relative scale; specifically, we used the percent change in flow or temperature in a given catchment relative to its control (i.e. [change in treatment post-harvest relative to control/pre-harvest] × 100) as a common response metric to compare across all catchments (Grant et al., 2008; Guillemette et al., 2005). This approach is intuitive for flow, which is challenging to compare on an absolute scale across different sized catchments; however, it is less ideal for temperature. Thus, we also report descriptive statistics about absolute changes in various temperature metrics to add context around these results. We also did not account for time post-harvest given that studies differed in duration and did not consistently report the timing of

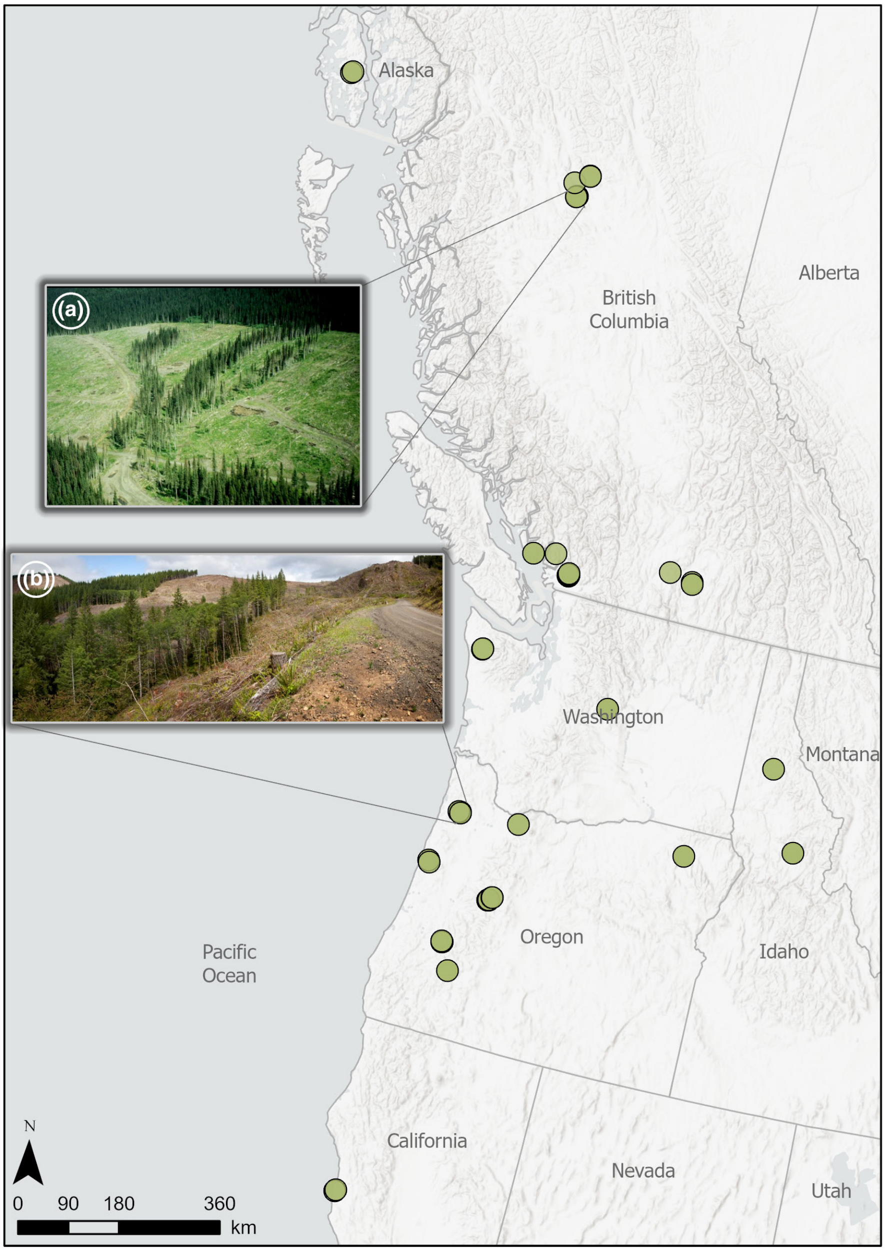

FIGURE 1 Map of catchment locations across Western North America. Insets show examples of paired catchment studies from the Baptiste (a) in British Columbia and the Trask (b) in Oregon [photos: D. Patterson (top) and D. Leer (bottom)].

impacts; thus, quantitative analysis was not possible. As a result, our approach does not consider whether impacts are permanent or transient. Instead, we focus on the maximum effect detected across the full duration of each study.

There were several instances of multiple studies examining forestry impacts in the same catchment. In some cases, this reflected distinct harvest experiments applied in a catchment over time; in others, it reflected re-analysis of the same experiments. We included the former case when quantifying the absolute ranges of response magnitudes; however, we only included unique catchments in any formal statistical analysis. For cases where multiple studies analysed responses to the same harvest treatments ($n=11$ for peak flow; $n=3$ for temperature), we extracted data from the most recent study, which had longer durations over which potential impacts could be detected. This decision did not influence the overall results, that is, more recent studies did not report consistently greater or lesser impacts.

For each catchment, we extracted information reported about harvest intensity, defined as the per cent of the catchment harvested, as well as catchment area and the method of harvest. Harvest methods have changed over time and varied considerably among studies, ranging from total clear-cutting that included the riparian area, retention of various types of riparian buffers and other selective harvest methods (e.g. thinning). Details about harvest methods were often limited; so, we used coarse groupings for analysis of 'clearcut' or 'other' for harvest type of the entire catchment, and 'present' or 'absent' for riparian buffers. We acknowledge this approach omits important details such as buffer width (Kiffney et al., 2004); however, it still captures broad differences among contrasting harvest methods. Harvest treatments in several studies were defined in reference only to riparian areas (Macdonald et al., 2003), but examination of historical photographs and communication with study, authors confirmed they could be categorized as 'clearcut' (H. Herunter personal communication).

We computed other potential watershed modifiers (e.g. lithology, hydrology and climate) using Geographic Information Systems (GIS). First, we extracted coordinates for flow and temperature locations. In some instances, the coordinates were reported directly in the studies; in others, we approximated them using Google Earth. While we have high confidence our approximations of catchment locations were accurate, small size (i.e. <1 km^2) and limited topographical variation in many of the catchments precluded our ability to delineate the watersheds from the coordinate point estimates, even from studies where coordinates were directly reported. Consequently, we were not able to extract local-scale watershed features such as slope, gradient or aspect. Instead, our analysis focused on broader scale biogeographical and climatic features derived from location estimates and described below. We used ArcGIS and ClimateWNA v. 7.31 (Wang et al., 2016) to extract these features from the point coordinates across all catchments. Further detail on the extraction process is described in Supporting Information 2. Here, we describe each watershed attribute and its hypothesized influence on responses to forestry.

1. *Mean annual precipitation* (*mm*)—Several studies have suggested that hydrological responses to forest disturbance differ with precipitation (Creed et al., 2014; Goeking & Tarboton, 2020). For example, Adams et al. (2012) suggested that streamflow in watersheds receiving less than ~500 mm/year may increase less or even decrease following forest disturbance relative to watersheds with greater precipitation levels.
2. *Precipitation as snow* (%)—Snow versus rain-dominated watersheds have hydrological differences that may affect their sensitivity to forestry. For example, Grant et al. (2008) suggested that peak flows in snow-dominated watersheds may respond more strongly to forestry relative to rain-dominated watersheds.
3. *Aridity*—Run-off responses to forest cover change have been shown to be less pronounced (or even reversed) in watersheds with higher aridity (Goeking & Tarboton, 2022). Thus, aridity may be expected to mediate forestry impacts on peak flows and potentially temperature. We calculated an aridity index for each catchment as the ratio of potential evaporative transpiration (PET) to mean annual precipitation averaged across the study years (see Supporting Information 2).
4. *Bedrock lithology composition* (*friability*)—Recent work has highlighted the importance of lithology as a key feature influencing watershed hydrology (Carlier et al., 2018) and responses to forest cover change (Bladon et al., 2018). Catchments with more friable lithology appear to have greater permeability and water storage capacity, thus could attenuate forestry impacts on flow and temperature relative to catchments with less permeable lithology. We extracted information on bedrock lithology (see Supporting Information 3) for each catchment point location. We classified catchments as 'permeable' if bedrock was classified as sedimentary, unconsolidated or quaternary (Bladon et al., 2018), and 'not permeable' for other bedrock types (e.g. igneous, volcanic). While this method is less precise than quantitative catchment-scale lithology composition (Bladon et al., 2018; Carlier et al., 2018), it likely captures differences among catchments across the broad spatial scale we investigated. Bedrock fracturing can also strongly influence catchment hydrology and likely responses to forestry (Hahm et al., 2019), but this information was not available for the watersheds we investigated, and is not included in our analysis.

In addition, we investigated the influence of elevation (m), latitude and longitude, which do not have clear mechanisms to modify forestry impacts, but act as broad proxies for other sources of watershed variation that could not be directly incorporated. We also incorporated the potentially modifying influence of catchment area; however, we limited inference into this effect given the constrained range of small catchments in the data set.

2.3 | Statistical analysis

All analysis was conducted in R Statistical Modelling Software (Version 4.2.2 R Core Team, 2023). We used simple descriptive statistics to examine the overall range of flow and temperature responses and their relationship to forestry intensity. First, we visualized overall response distributions with violin plots and report means, medians and standard deviations. We then used quantile regression to explore the overall response of flow and temperature to forestry intensity using the *quantreg* package in R (Koenker, 2009). For peak flow and temperature, we examined 5%, 25%, 50%, 75% and 95% quantile responses across the gradient of harvest intensity. These analyses are intended to be descriptive, thus we did not pursue formal statistical inference.

To investigate stressor–response relationships and watershed sensitivity, we constructed a series of statistical models to relate harvest intensity to peak flow and temperature responses and infer the relative influence of modifying watershed attributes. All models included harvest intensity (the per cent of a treatment watershed harvested), harvest mode ('clear-cut' or 'other') and catchment area. We then compared non-nested linear models with each of the potential modifying watershed covariates described above (section 2.2). We used Akaike's information criteria for small sample sizes (AIC_c) to identify the most parsimonious models for each response metric, defined by an AIC_c score under 2 (Burnham & Anderson, 2002). Each candidate model set included an intercept-only model without any predictor terms. Due to collinearity among variables (Figures S3 and S4), we did not examine interactions or multiple watershed covariates within the same model. All continuous predictor terms were centred to a mean of 0 and standardized by dividing by two standard deviations, while binary predictors (e.g. harvest mode and buffer presence) were left on original scale (Gelman, 2008). We fit models using least squares regression and examined underlying assumptions of normality and homogeneity of variance using diagnostic plots.

We inferred support for predictors in top-ranked models based on whether 95% confidence intervals around the coefficients overlapped zero. We then back-transformed predictors to examine effects on their original scales. We also examined means and 95% confidence intervals around the model intercepts, which approximate the statistical effect size of forestry treatments across all catchments, assuming average harvest intensity and watershed conditions. We interpreted 95% confidence intervals around the intercept not overlapping with 0 as an indication of a detectable effect of forestry at its mean value for a given metric.

Throughout all analyses, we assumed no a priori knowledge about the functional form of relationships, so we did not force regression intercepts through the origin despite it being conceptually intuitive to do so; that is, a response should in principle be zero when no stressor is applied. Our modelling is therefore better viewed as a test for linear stressor–response relationships over a specific range of forestry intensity (5%–100% harvest) as opposed to a comprehensive exploration of different stressor–response functional forms across the full stressor gradient (Jarvis et al., 2024).

3 | RESULTS

Forestry impacts were evident across all response metrics (Figure 2). Across all catchments and levels of harvest intensity, peak flows increased 19.9%, low flows decreased 26.3% and maximum temperatures increased 14.8% on average relative to control catchments. For temperature, these relative changes encompassed an average 2°C increase in max daily temperature, a 1.5°C increase in maximum 7-day average temperature, a 3.5°C increase in instantaneous maximum temperature and a 1.3°C increase in monthly max temperature (Figure 2d). There was significant variability around these effect sizes, with impacts on peak flows ranging from an 8% decrease to a 125% increase, low flow ranging from 0% to 50% decrease and temperature ranging from a 9.3% decrease to a 44% increase. 95% confidence intervals around model intercepts did not overlap 0 for both peak flow and temperature. Specifically, models suggest that peak flows would increase 22% (95% CI; 6.2%–38.5%) and maximum summer temperature would increase 22% (95% CI; 11.1%–32.1%) if the average harvest intensity levels were applied to a catchment with average conditions (e.g. area) across this data set.

Model selection indicated a clear top model for both response metrics, with only one model for each response with AIC$_c$ <2 (Table 1). For peak flows, the highest ranked model explained 26% of the variation and included an effect of latitude, in addition to harvest intensity, harvest mode and watershed size. For temperature, the top-ranked model explained 25% of the variation and included an effect of longitude. For both response metrics, our analysis revealed limited support for general stressor–response relationships, that is, the effect of harvest intensity (Figure 3). Specifically, both response metrics increased on average with increasing harvest intensity, but confidence intervals around these coefficients overlapped zero (Figure 4).

Based on 95% confidence intervals, our analysis indicated that peak flow and temperature responses to forestry were mediated by latitude and longitude respectively. For peak flow, latitude had

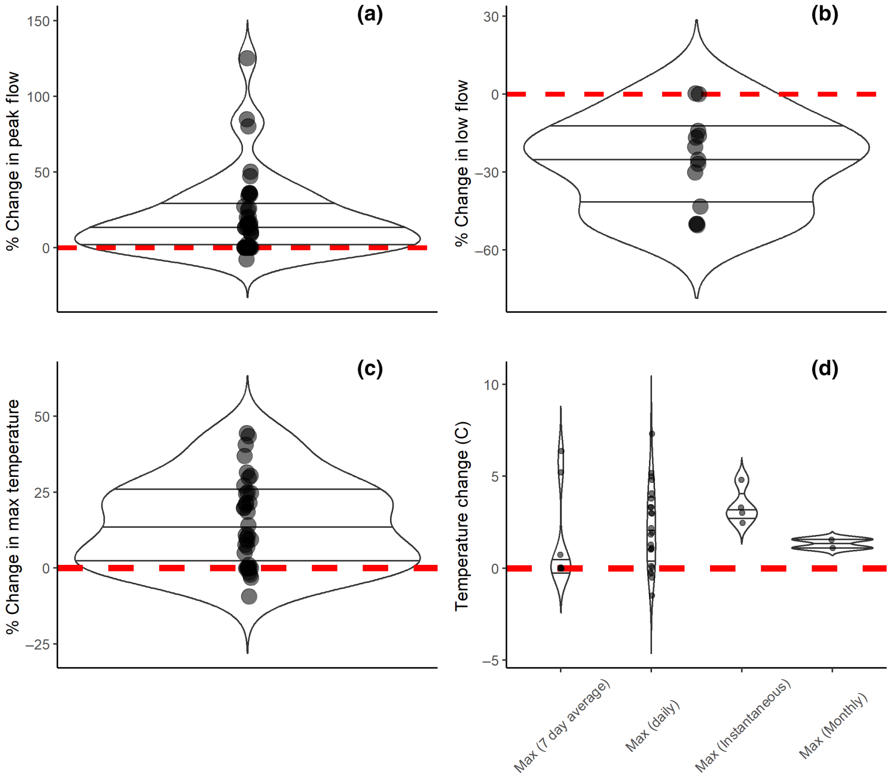

FIGURE 2 Violin plots showing the per cent change in peak flows, low flows and temperature (a–c) in a catchment relative to its control. Panel (d) shows absolute changes in temperature across different response metrics. The shape of the violin represents the kernel density around each data point and the three solid lines represent the 25th, 50th and 75th quantiles of the distribution. The horizontal dashed line at 0 indicates no change in a treatment relative to a control. Points are slightly jittered for clarity.

TABLE 1 Results of model selection analysis for peak flow and maximum summer temperature.

Model	AICc	ΔAICc	W	Log likelihood
Peak flow				
β(Harvest Intensity) + β(Area) + β(Harvest mode) + β(Latitude)	297.09	0.00	0.76	−140.80
Intercept only	299.82	2.72	0.19	−147.69
β(Harvest Intensity) + β(Area) + β(Harvest mode)	305.50	8.41	0.01	−146.55
β(Harvest Intensity) + β(Area) + β(Harvest mode) + β(% Snow)	306.03	8.94	0.01	−145.27
β(Harvest Intensity) + β(Area) + β(Harvest mode) + β(Lithology)	306.43	9.34	0.01	−145.47
β(Harvest Intensity) + β(Area) + β(Harvest mode) + β(Precipitation)	306.64	9.54	0.01	−145.57
β(Harvest Intensity) + β(Area) + β(Harvest mode) + β(Longitude)	307.47	10.37	0.00	−145.98
β(Harvest Intensity) + β(Area) + β(Harvest mode) + β(Elevation)	308.03	10.94	0.00	−146.27
β(Harvest Intensity) + β(Area) + β(Harvest mode) + β(Aridity)	308.56	11.47	0.00	−146.53
Max temperature				
β(Harvest Intensity) + β(Area) + β(Harvest mode) + β(Longitude)	286.69	0.00	0.75	−135.84
Intercept only	290.47	3.78	0.11	−143.05
β(Harvest Intensity) + β(Area) + β(Harvest mode) + β(Latitude)	292.27	5.58	0.05	−138.63
β(Harvest Intensity) + β(Area) + β(Harvest mode) + β(Elevation)	292.43	5.74	0.04	−138.71
β(Harvest Intensity) + β(Area) + β(Harvest mode) + β(Aridity)	293.04	6.36	0.03	−139.02
β(Harvest Intensity) + β(Area) + β(Harvest mode)	295.25	8.56	0.01	−141.59
β(Harvest Intensity) + β(Area) + β(Harvest mode) + β(Precipitation)	297.43	10.75	0.00	−141.22
β(Harvest Intensity) + β(Area) + β(Harvest mode) + β(Lithology)	297.47	10.79	0.00	−141.24
β(Harvest Intensity) + β(Area) + β(Harvest mode) + β(Buffer)	297.64	10.95	0.00	−141.32
β(Harvest Intensity) + β(Area) + β(Harvest mode) + β(% Snow)	298.18	11.49	0.00	−141.59

the largest standardized effect size (33.9; 95; 95% CIs: 13.5, 54.2). Forestry in higher latitude catchments was associated with greater increases in peak flows than lower latitude catchments. The top-ranked model suggested that if the average proportion of an average sized catchment was harvested, peak flow would increase 4.8% more than expected with every degree increase in latitude. For temperature, longitude had the largest standardized effect size (−18.9; 95% CIs: −30.2, −7.6). Forestry in higher longitude (i.e. more interior) catchments caused greater increases in temperature than lower longitude (i.e. more coastal) catchments, and the top-ranked model predicted temperatures would increase 2.8% less than expected with increasing longitude.

4 | DISCUSSION

Our synthesis of over half a century of paired catchment experiments across the North American range of Pacific salmon revealed that forestry generally increased peak flows, decreased low flows and increased maximum stream temperatures. The magnitude of these impacts varied considerably and there was minimal evidence for generalizable stressor–response relationships. Specifically, our analysis did not detect a relationship between harvest intensity and peak flow or temperature responses. We also found minimal support for any larger scale modulating factors (e.g. lithology, aridity or rain vs. snow hydrology) that we hypothesized would drive this variability and watershed sensitivity to forestry impacts. Taken together, these results suggest that the impacts of forestry can be substantial but unpredictably variable, overriding the detection of generalizable stressor–response relationships within the current existing body of experimental research.

Latitude and longitude were the only watershed features other than catchment area that appeared to influence sensitivity to forestry, with impacts on peak flow increasing northwards with latitude and impacts on temperature decreasing eastwards with longitude. In effect, our analysis showed that impacts vary across space, but could not detect what drives that variation. This suggests that the broad-scale physical and climate attributes we investigated may not have properly captured regional differences among watersheds. There may also be local-scale features that vary across regional gradients and more strongly mediate responses. These local-scale watershed features could include catchment slope, aspect, gradient and soil characteristics, which interactively influence the physical processes underlying flow and temperature (Buma & Livneh, 2017; Ellis et al., 2011; Poole & Berman, 2001).

Aspects of our methodology may have also obscured detection of generalizable stressor–response relationships and sensitivity heuristics. First, we were limited to coarse categorization of forestry practices and riparian buffer presence. Nuanced differences in forestry practices among studies may have a large bearing on outcomes. For example, the spatial configuration of harvest across sub-catchments of a watershed may strongly influence run-off responses

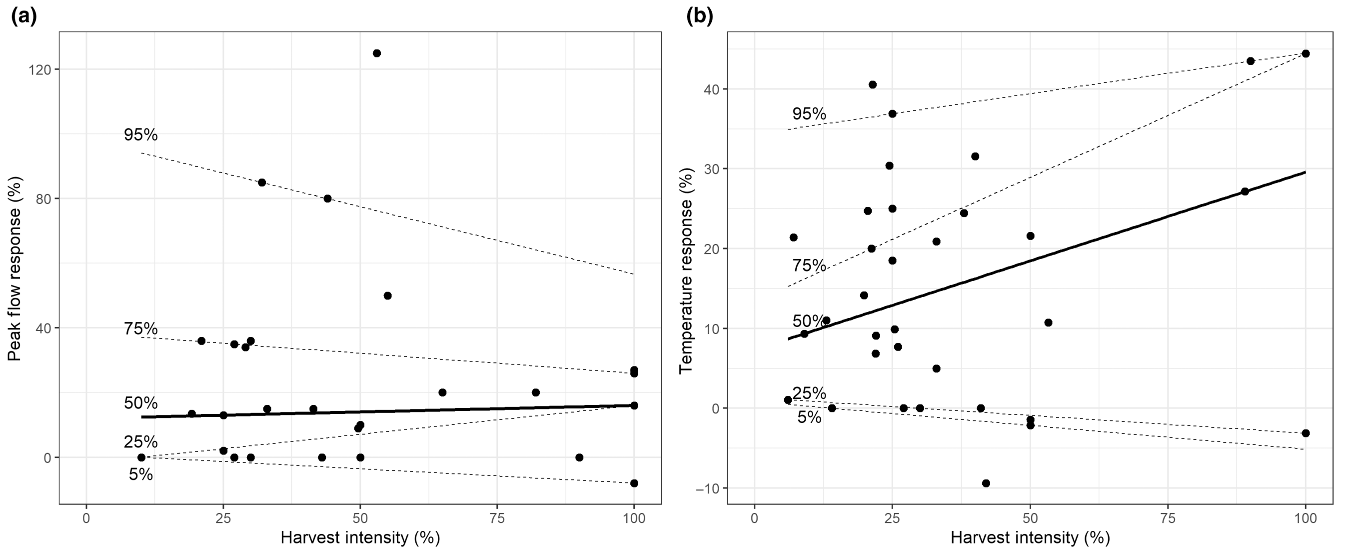

FIGURE 3 Relationships between harvest intensity (per cent catchment harvested) and peak flow (a) and maximum temperature (b) responses (the per cent change relative to control). Dashed lines represent different quantiles around the response. These lines were derived from univariate quantile regression relating forestry intensity to flow and temperature responses.

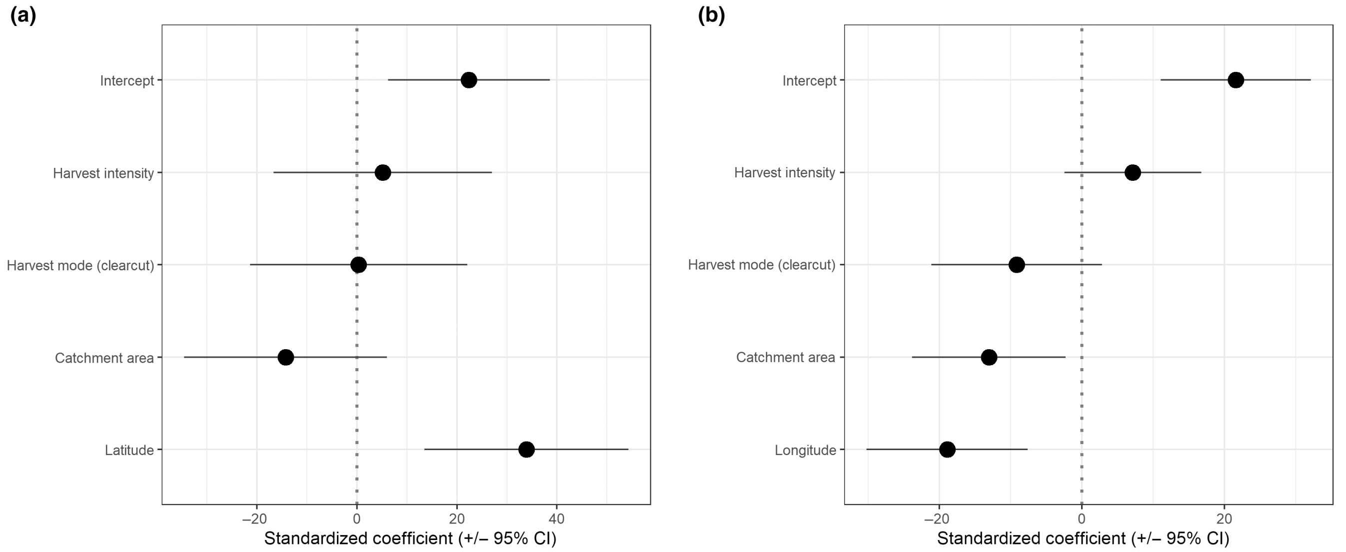

FIGURE 4 Standardized coefficient estimates for the most parsimonious statistical model explaining peak flow (a) and maximum temperature (b) responses to forestry. Lines indicate 95% confidence intervals around the estimate.

(Coble et al., 2020). Similarly, reliable information on pre-harvest forest attributes such as age and species composition was unavailable. Second, our peak flow analysis is complicated by differences in peak flow return periods among studies. In a previous synthesis of forestry impacts on peak flows, Grant et al. (2008) found that return period was an important response modifier, with the magnitude of impact declining with longer return periods. We were unable to explicitly account for return periods as they were not consistently reported and could not be approximated in many studies. Third, our inability to delineate watersheds reduced the precision of watershed modifier attributes, which may have obscured our analysis. Finally, despite the broad geographical scope of our search criteria, uneven clustering of studies (i.e. primarily coastal systems in Oregon) may have constrained the range of watershed modifiers and our reduced ability to detect effects. This final point likely explains the divergence between our results and other syntheses across broader geographical areas (Goeking & Tarboton, 2022).

It is also important to emphasize that our analysis focused on linear effects over the observed range of forestry intensity and did not explore non-linear functional forms due to limited sample size. Therefore, one interpretation of our results is that stressor–response relationships for forestry are non-linear and our models are incorrectly specified over the full range of forestry intensity. This is supported by the observation of positive intercept terms despite model intercepts being zero intuitively (i.e. the expected response when no stressor is applied). In effect, our results may indicate a threshold relationship increasing from 0% to around 5% harvest (the lowest harvest intensity level). However, we caution this interpretation without additional analyses, and instead emphasize the large variability across the much larger gradient of forestry intensity we observed.

Inferences from our results are also influenced by broader limitations of paired catchment studies. First, the majority of the catchments in these studies are small (i.e. $<1\,\text{km}^2$), and scaling results to larger systems is uncertain. While we accounted for catchment area in our analyses, we were limited by the range of catchment sizes represented in these studies. Forestry impacts occurring in a small subcatchment may be dampened or amplified over a larger watershed depending on the cumulative trajectories of both impacts and recovery in other constituent sub-catchments. For example, declining base flow due to regenerating vegetation decades after harvest (e.g. Gronsdahl et al., 2019) may be offset over larger scales if run-off is increasing in other stands or sub-catchments that have been harvested more recently (Coble et al., 2020). Alternatively, impacts may be amplified if stand ages are uniform over larger areas. Second, like all empirical studies our synthesis is retrospective. Consequently, forestry treatments in these experiments include a mix of historical and contemporary forestry practices. Altogether, these considerations suggest our results may represent the upper bounds of potential forestry impacts, especially to larger salmon-bearing watersheds. While these challenges underpin paired catchment studies, we do note that our results generally align with comparative approaches in salmon-bearing watersheds that include larger catchments (Bowling et al., 2000; Cunningham et al., 2023).

4.1 | Management implications

Our work has several important implications for management. First, it suggests that forestry can have strong impacts on flow and temperature but these impacts are highly variable and not predicted by generalizable stressor–response relationships across the range of watersheds we examined. More detailed studies and process-based modelling approaches may reduce this uncertainty to inform management (Fabris et al., 2018; Schnorbus & Alila, 2013), but they may not always be accessible given their information requirements. In situations where these risk assessment tools are not available, our results suggest forestry has the potential to have strong impacts across a wide range of harvest intensities.

Second, our study is among the first to quantify distributions of possible forestry impacts on salmon habitat across a large geographical area (but see Grant et al., 2008). Given the wide variability we observed, thresholds of forestry intensity developed in specific watersheds (Guillemette et al., 2005) may be poorly transferable to others. Planning and management in these information-poor situations may therefore be better informed by probabilistic distributions of risk or belief networks as opposed to static relationships or heuristics (Borgomeo et al., 2018; Pham & Alila, 2024; Ziemer et al., 1991). Thus, our results could directly inform these more holistic risk assessment approaches, which have further relevance in the context of cumulative effects (Tulloch et al., 2022).

Third, the general magnitude of flow and temperature change that we observed was within a range that can cause deleterious impacts to Pacific salmon. For example, several population model simulations have found reduced salmon spawner abundance resulting from peak flow and temperature increases within the range of our results (Battin et al., 2007; Nicol et al., 2021). Realized impacts to salmon are likely to be context-dependent, given that flow and temperature affect salmon through complex and non-linear pathways mediated by local geomorphology (Dralle et al., 2023; Gronsdahl et al., 2019; Sloat et al., 2017), behaviour and physiology (Railsback, 2021) and population dynamics (Ohlberger et al., 2018). While these complexities may further challenge our ability to clearly link forestry activities to salmon population trends (Peacock et al., 2023), our results still provide evidence that forestry poses a risk to salmon systems that in many cases already face an array of other pressures (Munsch et al., 2022).

Finally, our findings provide broader context around watershed management in a changing climate. Specifically, our results and others suggest that the magnitude of temperature and streamflow change from forestry may be comparable to projected temperature impacts from climate warming over the next century (Isaak et al., 2018; Mantua et al., 2010; Schnorbus et al., 2014). While direct quantitative comparison of these impacts is challenged by differences in scale, this is a critical observation for management. It suggests that past and ongoing forestry and other activities may substantially contribute to contemporary climate impacts and erode future climate resilience of watersheds (Moore & Schindler, 2022; Munsch et al., 2022). Yet, it also suggests that local management levers (i.e. forestry practices) will influence temperatures and streamflow in watersheds even as they are pressured by global climate warming (Fuller et al., 2022; Wondzell et al., 2019). Improving forest practices may therefore be an important opportunity to mitigate climate change impacts on Pacific salmon and other valued ecosystem components.

5 | CONCLUSIONS

Our synthesis revealed that forestry impacts on streamflow and temperature were consistently detectable, highlighting the risk forestry activities pose to Pacific salmon and aquatic ecosystems. At the same time, the magnitude of impacts was highly variable across different watersheds, highlighting challenges in developing generalizable predictions (i.e. stressor–response relationships). Forestry therefore represents a potentially significant risk, but one with high variability. While future work may uncover more nuanced stressor–response patterns and sensitivity modifiers (e.g. local watershed attributes), our findings emphasize the need for continued caution around assuming static thresholds or heuristics for management.

Watersheds are complex systems and our results reveal that forestry, a major activity, can have large but variable and unpredictable impacts. Thus, precisely predicting the impacts of human activities and being confident that activities carry no risk may not be scientifically defensible (Holling & Meffe, 1996; Schindler & Hilborn, 2015). However, synthesizing information on impacts can provide insights into the relative risks of different impact magnitudes, which can help guide key stakeholders and rightsholders to articulate the level of acceptable risk to guide decision-making.

AUTHOR CONTRIBUTIONS

All authors conceived the ideas and designed methodology; Sean M. Naman, Kara J. Pitman, Dylan S. Cunningham and Anna Potapova collected the data; Sean M. Naman and Kara J. Pitman analysed the data; Sean M. Naman led the writing. All authors contributed critically to the drafts and gave final approval for publication.

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CONFLICT OF INTEREST STATEMENT

The authors have no conflicts of interest to declare.

PEER REVIEW

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DATA AVAILABILITY STATEMENT

Data available from Dryad Digital Data Repository: https://doi.org/10.5061/dryad.xwdbrv1n2 (Naman et al., 2024).

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SUPPORTING INFORMATION

Additional supporting information can be found online in the Supporting Information section at the end of this article.

Figure S1. Histograms showing the distribution of attributes for catchments used in analysis of peak flow responses to forestry.

Figure S2. Histograms showing the distribution of attributes for catchments used in analysis of maximum summer temperature responses to forestry.

Figure S3. Correlation matrix for continuous watershed covariates used in analysis of peak flow responses to forestry. Colours indicate the strength of the correlation coefficient (*r*).

Figure S4. Correlation matrix for continuous watershed covariates used in analysis of maximum temperature responses to forestry. Colours indicate the strength of the correlation coefficient (*r*).

Table S1. Catchments used in analysis along with associated reference.

Supporting Information 3. Description of GIS methods.

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