

THE IMPACT OF TIMBER HARVESTING ON SALMON ABUNDANCE

William K. Jaeger
Department of Agricultural and Resource Economics,
213 Ballard Extension Hall
Oregon State University
Corvallis, OR 97331-3601
(541) 737-1419
wjaeger@oregonstate.edu

&

David L. Fluharty
School of Marine Affairs,
University of Washington
3707 Brooklyn Avenue N.E.
Seattle, WA 98105-6715
(206) 543-7004
fluharty@u.washington.edu

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Abstract

Although timber harvesting is widely believed to have contributed to the decline of wild anadromous salmonid populations throughout western North America, habitat alternations can have positive or negative effects on salmon that are often difficult to predict, and there exists no firm quantitative evidence of an overall negative influence. To address this lack of empirical evidence, the present analysis appraises the impact of timber harvesting on salmon abundance using a “cross-sectional” (many observations for a single time period) statistical analysis based on sixty-two coastal watersheds in Washington, Oregon, and British Columbia. Multiple regression analysis is used to evaluate the relationship between salmon abundance and geographical information system (GIS)-based measures of forest cover and disturbance. The analyses find a statistically significant positive association between the area of undisturbed forest and average salmon population size, indicating an increase of 0.6 salmon per hectare of undisturbed forest. No statistically significant effect is detected for the area of harvested forest land. Salmon abundance was found to be higher (lower) in Mesozoic (sedimentary) terranes, and it was found to increase at higher latitudes. These results lend quantitative support to the view that timber harvesting has had significant, long-term, detrimental impacts on salmon populations due to their impact on freshwater habitats.

Introduction

The populations of wild anadromous salmonids continue to decline throughout western North America due to human and natural factors including over-harvesting, hydropower operation, artificial propagation, climatic and oceanic changes, and the destruction and degradation of habitat through logging and other land-use changes (Nehlsen et al. 1991, Quigley and Arbelbide 1997). The relative importance of any one of these causes, however, is highly uncertain and subject to intense debate. For policy purposes, an improved understanding of these key relationships is urgently needed.

Concerning their freshwater habitats, a substantial volume of literature exists which as documented the aquatic requirements of salmonids at each life stage, the ways in which human activities have altered these watersheds (including changes in vegetation, soils, hydrology, sediment, temperature, nutrients, chemicals, and physical habitat structure), and the responses of salmonids to these changes (Meehan 1991; Quigley and Arbelbide 1997; Spence et al. 1996). Timber harvesting and associated road construction are widely believed to be an important cause of salmon decline in the Pacific Northwest (Nehlsen et al. 1991) due to the resulting disturbances in these freshwater habitats. These include increased rates of erosion and sedimentation (Quigley and Arbelbide 1997; US Forest Service 1993), changes in water quality and quantity, nutrient cycles, protective vegetation and dissolved oxygen (Chamberlain et al. 1991; Quigley and Arbelbide 1997), and physical changes in stream flow, temperature, fine sediment, woody debris, and channel morphology (Spence et al. 1996; Ralph et al. 1994; Halbert 1995).

However, habitat alterations can have either positive or negative effects that are often difficult to predict (National Academy of Sciences 1996), for example, increased light levels and invertebrate production can enhance overwinter survival and increase smolt size (Holtby and Scrivener 1989). Given the number of factors influenced by changes in streamside vegetation, and because interactions between such factors are still poorly understood, models of the impact of habitat change on salmon generally suffer from an inability to predict the consequences of interacting ecological processes (Mathur et al. 1985, Fausch et al. 1988).

Furthermore, preexisting empirical studies cannot be relied on to corroborate the inferred relationship: there exists no study for which statistically significant quantitative evidence has established the overall impact of timber harvesting on salmonid populations (Bisson 1992). Few controlled studies have attempted to measure this relationship. Two notable studies include the Alsea Watershed Study which compared the influence on streams of clear-cut and “patch-cut” watersheds with an unlogged watershed over 15 years, but was unable to determine the response of coho salmon (*Oncorhynchus kisutch*) because smolt populations decreased in all three watersheds (Hall et al. 1987). In a second study at Carnation Creek, British Columbia, the impacts of three streamside harvest treatments resulted in a sharp decline in steelhead (*O. mykiss*) smolts, but an increase in the number of coho smolts (Hicks et al. 1991; Hartman and Scrivener 1990), leaving the evidence inconclusive overall.

Non-experimental time-series studies have similarly been unsuccessful in confirming a relationship between logging and salmon abundance. Botkin et al. (1995) estimated species-specific escapement trends in the Rogue and Umpqua Rivers, and

found no detectable effect of changes in forest practices on returns of adult salmon (Botkin et al. 1995, p. 155). And Lee et al. (1997) employed “classification tree” models to associate the presence or absence of a given species with predictor variables such as biophysical environment and land management class in the interior Columbia Basin, but interpretation of their results is not straightforward and the authors acknowledge that their informal methods cannot be used to make strong inferences between watershed characteristics and species status (Lee et al., 1997).

Studies based on historical data for a single river or a small number of watersheds have generally been inconclusive for several reasons. Long-term abundance data have often relied on commercial and sport catches prone to error, escapement estimates often require conversion from spawner counts to estimated adult abundance (Neilson and Geen 1981), data may include year-to-year bias due to variation in conditions affecting stream visibility (e.g., weather, streamflow, etc.), and it may also be difficult to account for other temporal factors such as a rise in hatchery runs which may depress escapement of wild fish (Hilborn 1992). Collecting reliable historical data on the timing of land use and land cover changes can also be problematic (Halbert 1995).

Overall, the absence of successful empirical studies is due in part to the fact that long-term experimental studies of individual watersheds are costly and difficult to sustain; and non-experimental long-term studies face daunting obstacles given relatively short time-series of data exhibiting high variance and with multiple potential sources of bias. Nevertheless, and despite a widely-held belief that logging reduces salmon populations, the relationship has heretofore not been firmly supported with quantitative evidence, despite the importance of this relationship for public policy.

To overcome this deficiency, the present study has employed cross-sectional (across units of observation) statistical techniques. This method is widely used in the social sciences for understanding individual-, household-, or firm-level responsiveness to changes in institutional or economic variables, and it has also been used in ecology (Pickett 1989). Cross-sectional analysis may have advantages over time-series because sample size is not limited to the number of years of available data, and because the statistical power of the estimation will be greater from a sample of watersheds with highly variable land disturbances—in contrast to a single river exhibiting gradual changes.

Using data across watersheds for a single time period will avoid a number of problems associated with potential time-wise confounding factors (i.e., changes in salmon count methodology, changes in ocean conditions, cumulative impacts of harvest pressures, etc.), and it may be possible to control for other potential confounding factors—for example, using multiyear averages will smooth year-to-year stochastic elements in salmon escapement measures. Nevertheless, specification error due to the omission of a relevant variable is a potential source of bias, but this is true for any non-experimental empirical analysis whether cross-section or time series. Potential sources of misspecification error, and their likely consequences for the current analysis, are examined in detail in the Discussion section below.

Materials and methods

The current analysis employed cross-sectional multiple regression analysis for 62 watersheds in the coastal zones of British Columbia, Washington and Oregon to estimate

the relationship between salmon abundance and GIS-based measures of forest cover and disturbance, and including variables for geological types and latitude.

Data sets were compiled for both salmon abundance and forest characteristics. In the case of salmon abundance, data were assembled from official government sources (Washington and Oregon Departments of Fish and Wildlife, and British Columbia provincial agencies) by hydrological unit code (HUC), stream, index, estimation method, and year. Over 26,000 pieces of data were compiled representing over 500 streams between Alaska and California. From these, only data representing estimates of ‘total live fish,’ ‘population estimate,’ ‘total adults,’ or ‘run size’ were used in the analysis in order to avoid relying on partial estimations of the population such as ‘fish per mile’ or ‘pool counts.’ The set of useable data was much smaller than the full data set given the requirement that annual estimates of total run size be available to compute a five-year average for a common recent period. The period 1987-91 was chosen based on data availability. Alaska and California were eliminated from the analysis because complete data were available from only three rivers in Alaska and only one river in California. The data set contained only 13 Alaskan rivers where the salmon counts were estimates of total run size for the time period 1987-91. Of these, only three rivers appeared in the forest condition data set.

Data on coastal forest condition were assembled in two separate data sets, and were compiled for the current study by Inforain/Ecotrust, a non-profit GIS organization based in Portland, Oregon. First, using existing estimates of coast temperate rainforest (CTRF) disturbances, Inforain compiled databases from GIS sources which produced basin-level data on a) total watershed area, b) area of CTRF, c) barren areas, and d) area

of ‘developed’ or disturbed CTRF. CTRF data for 986 watersheds were extracted from a database developed to produce other publications (see Schoonmaker et al. 1997). The area within each watershed identified as undisturbed CTRF was based on photographic and aerial inspection, and ground-truthing based on comparisons with known CTRF areas.

The second data set on disturbance of forests in coastal watersheds compiled GIS-based estimates of forested area by seral stage. Forest seral stages were available in three classifications: early (between 10% and less than 70% crown cover), middle seral stage (crown cover greater than 70%, less than 10 % of crown cover in trees greater than or equal to 21 inches diameter-breast-height (dbh)), and late seral stage (crown cover greater than 70%, and more than 10% of crown cover in trees greater than 21 in. dbh). These data included 553 watersheds in British Columbia, Washington, Oregon, and northern California.

The overlap of valid data for both land conditions and salmon abundance left a combined data set containing only 62 watersheds distributed along coastal British Columbia, Washington and Oregon (see Figure 1). The number of observations for each species among the 62 watersheds was 37 for chum (*O. keta*), 16 for steelhead (*O. mykiss*), 10 for chinook (*O. tshawytscha*), 7 for coho, 7 for pink (*O. gorbuscha*), and 5 for sockeye (*O. nerka*).

The small number of observations for each individual species makes it unfeasible to estimate species-specific versions of the model. Nevertheless, this may not represent a limitation, however, since a model to estimate the abundance of one species as its dependent variable will be misspecified unless the abundance of other species are

included as independent variables. Obviously, however, the abundance of other species and their relationship with timber harvests are not independent, so that a relationship between the abundance of one species and forest practices will likely reflect incomplete measures of the cumulative effects of timber harvesting on fish populations since compensating adjustments among species are likely to occur (Bisson et al. 1992). For example, one Oregon study found that diversity of fish assemblages in streams in basins with low timber harvest levels was greater than in streams in basins with high timber harvest levels (Reeves et al. 1993). That being the case, models based on data for a single-species will be misspecified and would produce biased estimates of the coefficients as well as inaccurate inferences of significance due to the omission of the populations of other species as relevant covariates in the model.

Thus, the hypothesis being tested here is whether timber harvesting has an adverse *overall* effect on the abundance of salmon. Finer distinctions regarding effects on the relative abundance of different species are interesting questions. For example, the species composition may be affected by habitat changes without impacting the overall fish density (Bisson et al. 1992). However, these questions are beyond the scope of the current study given the limited data and sample size. Indeed, the potential success of the analysis is enhanced by focusing on aggregate anadromous fish density, precisely because it avoids the misspecification errors that would arise in species-specific models. There remains, however, the possibility that differences in the species composition among the 62 streams in our sample may have introduced a specification error. This possibility is tested for and rejected as reported below.

For the 62 valid observations, two additional variables were added to the data set in order to account for some of the physical and biological variability in the region that influences the suitability of these freshwater habitats. First, the latitude of each watershed's river mouth was introduced to act as a proxy for north-to-south differences in biophysical characteristics. Second, the major geological characteristics of each watershed were identified in order to reflect differences in importance geological and geomorphological processes that influence ecosystem structure and dynamics. Based on Montgomery (1997), each watershed was placed in one of three categories; a) Mesozoic accreted terranes consisting of partially to highly metamorphosed sedimentary and igneous rocks, b) granitic batholiths (Cretaceous and Tertiary) comprising the uplifted and eroded roots of volcanoes; and c) Tertiary sedimentary wedges consisting of marine sedimentary and volcanic rocks. The rock types associated with the sample basins shown in Figure 1 can be differentiated as follows: coastal Washington and Oregon (sedimentary), mainland British Columbia (granitic), and Vancouver Island (Mesozoic). Additional descriptive statistics for the data set are presented in Table 1.

Linear regressions were estimated for a range of reduced-form models where the dependent variable was average salmon abundance for 1987-91. Average salmon abundance over a five-year period is used so that annual variations from factors such as ocean conditions and measurement error will be greatly reduced. The usual linear regression model takes the form of

$$Y = \alpha + \sum_i \beta_i X_i + \varepsilon$$

where \mathbf{Y} is a $T \times 1$ vector of observed or estimated variables (average salmon abundance for the 62 streams), \mathbf{X} is a $T \times K$ matrix of exogenous variables, and $\boldsymbol{\beta}$ is a $K \times 1$ vector of coefficients. The intercept term, α , has been constrained to equal zero for these models implying only that it is reasonable to expect that as the size of the watershed goes to zero, salmon abundance will go to zero as well. The elements of $\boldsymbol{\varepsilon}$ are independent random variables with the usual assumptions.

The independent variables considered in the current model are categorized as: total basin area, \mathbf{A} ; forested area \mathbf{F} ; forest area by CTRF category (undeveloped and developed) \mathbf{X}^D ; seral stage forested area (early, mid- and late seral stage, combined early- and mid-seral stage), \mathbf{X}^S ; rock type (Mesozoic accreted terranes, granitic batholiths, sedimentary), \mathbf{X}^G ; and latitude, \mathbf{L} . All size and forest variables are in hectares. The rock type variables, \mathbf{X}^G , are binary or “dummy” variables taking values of 1 or 0 for “present” or “absent,” respectively. In the current setting, the effect of rock type on salmon abundance is expected to depend on watershed size. We therefore introduce them as interaction terms by forming the product of the binary variable and other area-dimensioned variables.

Various specific forms of the general model have been estimated with different combinations of the independent variables in order to consider a range of plausible specifications. Not all variables are included in each model, especially where doing so would introduce redundancy or extreme multicollinearity (e.g., including total forest, developed forest, and non-developed forest, all in the same model). The models differ primarily in terms of the specific interactions between binary variables and forest/area variables. The three model types for analysis of the CTRF forest disturbance data are:

$$\mathbf{Y} = \sum_i \beta_i^{AG} \mathbf{A} \mathbf{X}_i^G + \beta^F \mathbf{F} + \sum_j \beta_j^D \mathbf{X}_j^D + \beta^L \mathbf{L} + \varepsilon \quad (\text{Equation 1})$$

$$\mathbf{Y} = \sum_j \beta_j^D \mathbf{X}_j^D + \sum_i \beta_i^{FG} \mathbf{F} \mathbf{X}_i^G + \beta^L \mathbf{L} + \varepsilon \quad (\text{Equation 2})$$

$$\mathbf{Y} = \beta^F \mathbf{F} + \sum_i \sum_j \beta_{ij}^{DG} \mathbf{X}_j^D \mathbf{X}_i^G + \beta^L \mathbf{L} + \varepsilon. \quad (\text{Equation 3})$$

For the seral stage data set, the two model types estimated are:

$$\mathbf{Y} = \sum_i \beta_i^{AG} \mathbf{A} \mathbf{X}_i^G + \sum_j \beta_j^S \mathbf{X}_j^S + \beta^L \mathbf{L} + \varepsilon \quad (\text{Equation 4})$$

$$\mathbf{Y} = \beta^F \mathbf{F} + \sum_i \sum_j \beta_{ij}^{SG} \mathbf{X}_j^S \mathbf{X}_i^G + \beta^L \mathbf{L} + \varepsilon. \quad (\text{Equation 5})$$

Each of these five models was also estimated with the addition of species variables in order to test for species-specific effects, or for the possibility of a species-related specification bias on the overall results.

Results

The expected results for the analyses included finding that: a) salmon are more abundant in large watersheds than small, with salmon abundance diminishing to zero as watershed size goes to zero; b) undisturbed or late seral stage forests are more favorable to salmon than developed or early seral stage forests; c) abundance of salmon is highest in primarily Mesozoic watersheds, and lowest in primarily sedimentary watersheds, and d) salmon are more abundant at relatively high latitudes across the zone being studied.

An explanation for these anticipated relationships are presented in the discussion section below.

Results for versions of equations 1 through 5 are presented in Tables 2 and 3 for the CTRF and seral stage data, respectively. Due to the high degree of heteroscedasticity detected in the data (based on highly significant Breusch-Pagan tests), White's robust covariance matrix method (White 1982) was used for all regression results reported. F tests for each model indicate they are all highly significant (above the 0.001 level). Nearly all the results are consistent with our expectations. The principal results confirm that salmon abundance is positively associated with undisturbed forests, and forests in late seral stage, with very similar estimated coefficients of 0.67 and 0.60 for the two data sets in models 1A and 4A. These coefficients suggest an increase of approximately two salmon for every additional three hectares of undisturbed forest. No statistically significant effect is detected for the area of developed forest land.

Regression results involving the seral stage forest database did not produce statistically significant coefficients for the forest area variables when early, mid- and late seral stage areas were introduced separately. However, when early and mid-seral stage data were aggregated, the distinction between early-middle versus late seral stage resulted in statistically significant coefficients in the models, and coefficient values similar to the results using the other database. The inability of the analysis to distinguish between early and mid-seral stage forests is not surprising since the long-term effects of logging on salmon abundance, and the full recovery of the riverine habitat, are not likely to occur between early and mid-seral stages.

The results indicate a strong effect of geological characteristics. Estimates from both data sets indicate Mesozoic accreted terranes to be most favorable to salmon, followed by granitic batholiths, while Tertiary sedimentary zones were least favorable.

The estimated relationship for Mesozoic terranes appears to be as strong as for undisturbed forest, suggesting an additional two salmon for every three hectares of Mesozoic terrain.

Models 3 and 5 include interactions between forest condition and rock type which, in some cases, produced seemingly counterintuitive coefficients. For example, in model 3A the coefficients on developed forest area are generally negative and significant (especially for granitic and sedimentary terrain), and we see positive coefficients for non-developed forest areas of all rock types. The estimated coefficient for Mesozoic terranes is positive for both developed and non-developed forest areas, which might lead to the suggestion that disturbed forests on Mesozoic terranes are good for salmon. However, the estimated coefficients for developed forests from models 3A and 3C (1.218, 1.19) are not statistically distinguishable from the corresponding coefficients on non-developed terranes (0.96, 0.95). The associated t-values for this comparison are 0.86 and 0.91 respectively. These results suggest that forests on Mesozoic terranes are advantageous to salmon, but that we were unable to detect a differential effect between developed and non-developed forests in this particular case.

In the case of the seral stage specification in model 5, similar results suggest that forests on Mesozoic terranes are highly advantageous to salmon compared to other rock types, and that this holds whether they are in early, mid- or late seral stage. For example in model 5B, the coefficient on total forest area is positive and strongly significant (0.836). Since the binary variable when forests are on Mesozoic terranes have been omitted in this specification (this is necessary to avoid a binary variable singular matrix problem), the coefficient on total forest area can be interpreted as the estimated

relationship with Mesozoic forests (disturbed or undisturbed). Elsewhere for this model we see that all the coefficients on forest condition (by geologic type) are negative and significant. However, given this particular functional form, these results must be interpreted as representing deviations from the “base case” rock type (the one for the omitted binary variable—Mesozoic). In model 5B, then, the “net effect” of either granitic or sedimentary forests is computed by summing the two coefficients. In the case of sedimentary forests in late seral stage, for example, we subtract 0.694 from 0.836 to get a positive 0.142. In this way, we may note that with model 5B, the ‘net coefficients’ for granitic and sedimentary forests are all positive for late seral stage and negative for early seral stage, as we would expect. In the case of Model 5A (where the binary variable for forests on granitic terranes was omitted), the same is true for sedimentary forests, but in the case of Mesozoic forests this result for the ‘net effect’ suggests that Mesozoic forests are favorable to salmon in either seral stage category.

Salmon abundance was also found to be higher at higher latitudes, with positive coefficients in all regressions ranging from 84 to 176. The coefficients were statistically significant for all the seral stage regression models, but only for one of the CTRF regressions.

Each of the models reported in Tables 2 and 3 was re-estimated with the addition of species-specific binary variables (1/0 for present/absent). None of these binary variables was found to be statistically significant, and changes in the estimated coefficients compared to those reported in Tables 2 and 3 were negligible. Detailed results for these models are therefore not reported.

Discussion

This study was undertaken with two primary objectives. First, to overcome the lack of empirical evidence for understanding the impact of timber harvesting on salmon abundance; and second, to test the potential value of a cross-sectional or “space-for-time” substitution approach in this context. In regard to the first objective, the analysis has generated statistically significant quantitative evidence showing a negative relationship between forest disturbance and salmon abundance in the region under study. These results represent a confirmation of the widely-held belief that timber harvesting has significant detrimental impacts on salmon populations.

Given the highly aggregate nature of the data, the present analysis cannot detail the specific mechanisms or processes by which this reduction in salmon populations has occurred. These results, however, reinforce and are supported by findings from prior studies focused on the individual mechanisms at work. Preexisting studies have identified factors including reductions in large woody debris, increases in water temperature, and increased erosion and sedimentation which has been shown to impair the reproductive success of salmon due to increased turbidity, loss of interstitial spaces for use by juveniles, the smothering of eggs by fine sediments, loss of deep pools, and blockage of spawning habitat by landslides (see National Academy of Sciences, 1996, and U.S Forest Service, 1993, for summaries of this evidence). Direct support of the impact of logging on particular aspects of salmonids lifecycles comes from evidence that coho smolt production declined during early post-logging cycles compared with pre-logging, and that pre-smolt fry emigrants increased (Hall et al. 1987), and based on the observation that logging led to earlier seaward migration of coho smolts, a reduction in ocean survival and

recruitment has been inferred (Holtby and Scriverner 1989; Hall et al. 1987). Taken together, the current analysis and preexisting studies are mutually reinforcing: the preexisting literature on individual mechanisms provide a strong basis for deducing which underlying processes have led to the aggregate evidence captured in the current analysis; and the current analysis provides much-needed corroborative evidence that the cumulative effect of these individual mechanisms has caused overall reductions in abundance of the affected stocks.

For example, the existing literature provides a basis for speculation about nuances in the current findings which may not be obvious. For example, the results in Table 3 are somewhat equivocal—at least in the case of Mesozoic terranes—suggesting that early or mid-seral stage forests may actually be beneficial to salmon. This is not entirely incompatible with the existing theoretical and empirical literatures which suggest that salmonid productivity may be enhanced temporarily by increased light levels and invertebrate production (Holtby and Scriverner 1989).

Salmon abundance was found to be higher in watersheds with Mesozoic accreted terranes as compared to granitic types, and lowest in sedimentary terranes, consistent with expectations. The characteristics of catchment rock type which may influence salmon abundance include their effect on water quality, the abundance of well-aerated gravel for spawning, and the timing of runoff during specific times of the year. Lithology can affect water quality by controlling landslide frequency and surface erosion of sediment, both of which influence channel stability and stream turbidity. Fine-grained marine rocks, weathered volcanic rock, and some deeply weathered granitic rocks, for instance, tend to erode rapidly, producing large loads of fine sediment, relatively unstable

channel beds and banks and only minor amounts of gravel. In contrast, unweathered intrusive and fresh volcanic rocks and high-grade metamorphic rocks generally are less prone to failure by landslides and erode to produce resistant gravel and coarse sand (David P. Dethier, Department of Geosciences, Williams College, Williamstown, MA 01267, personal communication).

Thus, among the three generalized rock types represented in the study area, the Mesozoic accreted terranes are typically characterized by low amounts of suspended sediment, stable channels and abundant resistant gravel. Catchments that drain granitic batholiths are likely to display variable sediment and channel characteristics dependent on subtle differences in rock chemistry and the degree of rock weathering. And catchments underlain by Quaternary sediments, volcanic rocks or Tertiary sedimentary rocks are likely to produce excess amounts of fine sediment and relatively minor amounts of resistant gravel (David P. Dethier, Department of Geosciences, Williams College, Williamstown, MA 01267, personal communication).

The strong positive effect on salmon abundance for Mesozoic forest areas, but independent of forest condition, represents one empirical result that was not anticipated. A possible explanation for this may be that high-grade metamorphic rocks do not weather rapidly, and that timber harvesting and associated road construction do not appreciably alter either the manner in which they erode to produce resistant gravel and coarse sand, or their susceptibility to landslides. The tentative inference that one might draw from this evidence is that timber harvests on these particular Mesozoic terranes are less harmful to salmon than for other rock types.

Higher salmon abundance was also found to be associated with higher latitude across the region under study, a result consistent with the anticipated effects arising from differences in coastal and ocean conditions. Watersheds in the relatively northerly zone offer conditions more favorable to anadromous for at least two reasons. First, the oceanic boundary between the cool, nutrient-rich northern currents and warm, nutrient-poor southern currents occurs off the coast of Washington, Oregon, and northern California, descending infrequently to the southerly boundary of the zone (as far as northern California on an average of 1 in 4 years)(US Forest Service 1993). Thus, favorable ocean conditions occur more frequently off the coast of British Columbia, and less frequently moving to lower latitudes. Since survival rates for at least some stocks will be higher when ocean conditions are favorable (Nickelson 1986), this relationship would contribute to higher salmon abundance in the more northerly watersheds in our sample.

Second, the shoreline-to-coastline ratio is higher in the north, rising from approximately 5, to 20, to 27, in Oregon, Washington, and British Columbia, respectively (Bottom et al. 1986). A higher shoreline-to-coastline ratio reflects increased number of bays, estuaries, and other near-shore rearing areas, and these have been found to be important to the future ocean survival of anadromous fish (see US Forest Service 1993 for a summary of evidence). In addition to these differences, latitude may also be serving as a proxy for other anthropogenic influences, given that human population density varies inversely with latitude over this range.

Thus, our aggregate analysis has produced results which are consistent with, and corroborate, preexisting evidence regarding specific underlying processes—while not able to independently pinpoint the specific mechanisms at work.

Turning to the question of specification error, we are interested in the potential effects of omitting a relevant explanatory variable from the estimated model, especially a variable which may have led to a Type I error. Since not all potentially relevant variables have been included in our model, the estimated coefficient on the included explanatory variable will be biased if an omitted variable is correlated with the explanatory variable; there will be no bias, however, if the two variables are uncorrelated. The direction of the bias will be positive if the correlation between the two variables, and the underlying relationship with the omitted variable, both have the same sign. This would produce a Type I error. If the two variables are uncorrelated, no bias in the estimated coefficient will result (Kmenta 1971).

Even if an omitted variable (which influences the dependent variable) is uncorrelated with the included explanatory variable, the estimator of the variance of the coefficient for the explanatory variable will be higher than would be the case if the omitted variable were included. This will lead to unduly conservative conclusions for the usual tests of significance and confidence intervals: it will increase the probability of a Type II error, making it less likely that a valid relationship between timber harvesting and salmon abundance will be detected based on standard significance tests (Kmenta 1971).

We want to consider the possibility that our results reflect a Type I error with respect to three variables which, while expected to affect salmon abundance, are omitted due to lack of available data. These are ocean conditions, salmon harvest pressures, natural forest disturbances. The issue of species composition is also considered.

First, ocean conditions are clearly correlated with salmon abundance across the sample of watersheds for the 1987-91 period, with the influence likely to vary from north

to south as suggested by our results for latitude (discussed above). There is, however, no reason to expect that these differences are correlated with timber harvesting. Thus, we have no reason to believe that the estimated coefficient is biased, but we can conclude that the levels of significance for the estimated coefficients in Tables 2 and 3 are more conservative than necessary due to the upward bias on the estimated variance of the coefficient.

Second, the influence of salmon harvest pressures on stock abundance may be correlated with timber harvests to the extent that fishers, and fishery managers, are able to concentrate harvests on relatively healthy stocks. This may occur when fishing effort is concentrated at a river's mouth as salmon return to spawn. If this were the case, the area of undeveloped and late seral stage forest will be correlated with harvest pressures. Since harvest pressure will have a negative influence on salmon abundance, this source of specification error will introduce a negative bias in the estimated coefficient for undeveloped and late seral stage forests, raising the possibility of a Type II error. Thus our results may actually *understate* the effects of timber harvesting on salmon abundance due to this effect. (Note that a similar analysis based on time series data for a single river would likely involve data for which undisturbed forest area and harvest pressures are negatively correlated over time, creating a positive bias in the estimated coefficient and raising the probability of a Type I error.)

A third potential source of bias concerns the inclusion of both anthropogenic disturbances and natural disturbances in the data. Unlike omitted variable bias, this will be a form of measurement error which may lead to confounding the two influences. The coastal watersheds being studied are very prone to slumps and earth flows, a type of

natural disturbance which may have deleterious effects in the short- and medium-term, but which are recognized to be beneficial to the long-term productivity of watersheds (Reeves et al. 1995). For large disturbances such as windthrows and wildfires, evidence suggests that it may take 10 years or more for salmon populations to return to the normal range of predisturbance abundance (National Academy of Sciences 1996). The only large disturbance relevant to our sample, the 1962 Columbus Day storm, can reasonably be ignored because it preceded our sampling period by 25 years.

Nevertheless, if the forest areas affected by slumps and earth flows were included in our measure of anthropogenic disturbances (logged or early- and mid-seral stage forest area), then this might confound the interpretation of the results if their aggregate effects are significant and if the short- or medium-term effects of this kind of disturbance on salmon are quantitatively different from the effects of anthropogenic disturbances. Fortunately for our purposes, these disturbances tend to be much too small (typically 15 – 20 acres) to be detected in our data which has a spatial resolution of 250 acres (Peter Schoonmaker, PO Box 9127, Portland, Oregon 97207, personal communication, July 6, 2000). These natural disturbances therefore represent another omitted variable which, like ocean conditions, is highly unlikely to be correlated with timber harvesting, but which raise the probability of a Type II error.

Finally, as mentioned above, there is a possibility that differences in the species composition among the 62 streams in our sample have introduced a specification error. Among the streams in our sample there may be natural variations in the species mix across streams (due to differences in habitat), but there is no reason to expect that this natural variation is correlated with timber harvesting. In addition to natural variation,

there may be variation in species composition caused by timber harvesting. In this case, timber harvesting could produce increases or decreases in the populations of specific species compared to their numbers in an undisturbed watershed. However, to estimate the overall affect of timber harvesting on salmon abundance abstracting from these fine distinctions is appropriate. Although the most likely possibility is that the omission of species variables will produce only a Type II error, since we have data on the species represented in our sample, we can include them in the model to test for evidence of their effects directly.

As mentioned above, each of the five models represented in Tables 2 and 3 was re-estimated with the addition of species-specific binary variables for each of the species present in the sample. Evidence in the data that variations in salmon abundance are due to differences in their species composition should be detectable with these variables, leaving non-species specific sources of variation to influence the estimated relationship with forest condition variables. However, the results for these estimations with binary species variables were not appreciably altered; none of the species variables was found to be statistically significant, and changes in the estimated coefficients on the forest condition variables compared to those reported in Tables 2 and 3 were nil. We, therefore, find no apparent evidence from our sample to suggest that differences in species composition have given rise to a Type I error in our results. This should not be interpreted, however, as a positive finding of uniformity among species in their response to timber harvesting.

In regard to the second objective of the study—to test the potential value of using cross-sectional statistical methods in this context—the results offer encouragement that analysis using cross-section data or space-for-time substitutions have potential for

overcoming some of the limitations associated both with controlled studies and non-experimental time series analysis. In an environment characterized by high variability and numerous sources of variation, these results indicate that a sample of only 62 observations was sufficient to substantiate the negative impact of timber harvesting on salmon abundance, an outcome indicative of the potential value of the technique.

The implications for future research suggest that studies of this kind may be successful, especially given larger samples and more detailed data on watershed characteristics and types of anthropogenic disturbance. For example, road building and road density are generally believed to have an adverse impact on salmon – although these variables are highly correlated with forest disturbance since most road building in these areas is associated with logging. More sophisticated models including measures of topography, or combined measures of forest condition by type of topography, distance to stream, etc. may also be successful. The recent availability of high resolution data on forest cover and land use characteristics for large areas of the coastal Pacific Northwest has created new opportunities, making it possible to derive detailed land cover and riparian buffer characteristics for specific watersheds and river reaches over large geographical area.

More comprehensive and detailed studies of this kind should be undertaken. The need for a better understanding of these relationships extends beyond regulating forest practices. Given the cyclical nature of ocean conditions and their effect on salmon survival, the protection and restoration of freshwater habitats is increasingly seen as essential in order to protect salmon populations during periods of low ocean survival. The scientific and policy problem is this: The cyclical nature of salmon abundance arising

from fluctuations in the ocean environment may make it difficult to sustain public and policymaker support for long-term efforts to protect and restore freshwater habitats (Lawson 1993). The evidence presented here lends credibility to the concern that continued habitat degradation combined with the cyclical ocean survival conditions may lead to unanticipated declines or extinctions.

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Table 1. Descriptive statistics

Variable	Units	Mean	Range
Salmon abundance	total live fish	20,288	8 – 212,394
Basin area	hectares	55,409	3,157 – 560,135
Total forested area	hectares	33,950	3,158 – 139,872
Undisturbed forest (CTRF)	hectares	11,023	0 – 76,230
Early seral stage forest	hectares	5,502	0 – 32,699
Middle seral stage forest	hectares	7,980	0 – 39,965
Late seral stage forest	hectares	17,217	0 – 88,235
Latitude	degrees	48.2	45.1 – 50.1
Mesozoic terranes	binary (1/0)	0.242	0 - 1
Granitic terranes	binary (1/0)	0.338	0 - 1
Sedimentary terranes	binary (1/0)	0.419	0 - 1

Table 2. Regression results when CTRF forest condition is regressed on average salmon abundance (1987-91) for nine alternative model specifications.

	<u>Model 1A</u>	<u>Model 1B</u>	<u>Model 2A</u>	<u>Model 2B</u>
Total forest area		0.272 * (0.155)		
Non-developed forest	0.672 * (0.375)			0.717 (0.690)
Developed forest	0.024 (0.273)		-0.717 (0.690)	
<u>Forest area by rock type</u>				
Mesozoic:			1.608 * (0.809)	0.891 *** (0.286)
granitic:			0.672 * (0.396)	-0.448 (0.342)
sedimentary:			0.567 (0.616)	-0.150 (0.133)
Total basin area			0.016 (0.053)	0.016 (0.053)
<u>Basin area by rock type</u>				
Mesozoic:	0.662 * (0.362)	0.601 * (0.316)		
granitic:	0.015 (0.053)	0.341 (0.060)		
sedimentary:	-0.129 (0.166)	-0.243 * (0.139)		
Latitude	154.000 (99.5)	171.000 * (101.0)	149.000 (97.7)	149.000 (97.7)
R-squared	0.360	0.346	0.399	0.399

Notes: Standard deviations are in parenthesis. Significant t-tests are indicated as * = 0.1 level; ** = .05 level; *** = 0.01 level. R-squared values are adjusted to account for degrees of freedom; nevertheless they cannot be interpreted in the usual way due to the omission of the intercept term. F tests indicate that all models are significant at the 0.001 level.

Table 2. (Continued)

	<u>Model 3A</u>	<u>Model 3B</u>	<u>Model 3C</u>	<u>Model 3D</u>
Total forest area		1.153 *** (0.359)	0.035 (0.097)	1.305 ** (0.542)
<u>Non-developed forest area by:</u>				
Mesozoic:	0.964 (1.069)		0.946 (1.019)	
granitic:	2.204 *** (0.436)	1.063 ** (0.531)	2.192 *** (0.448)	
sedimentary:	0.594 * (0.338)	-0.560 (0.496)		
<u>Developed forest area by:</u>				
Mesozoic:	1.218 *** (0.242)		1.190 *** (0.226)	-0.198 (0.473)
granitic:	-1.463 *** (0.348)	-2.619 *** (0.495)	-1.504 *** (0.343)	-2.025 ** (0.972)
sedimentary:	-0.085 (0.104)	-1.232 *** (0.326)		-1.546 ** (0.596)
Latitude	96.983 (99.3)	89.957 (86.0)	84.420 (97.6)	142.053 (97.7)
R-squared	0.508	0.516	0.511	0.480

Table 3. Regression results when seral stage forest condition is regressed on average salmon abundance (1987-91) for five alternative model specifications.

	<u>Model 4A</u>	<u>Model 5A</u>	<u>Model 5B</u>	<u>Model 5C</u>
Total forest area		0.187 *	0.836 ***	-0.034
		(0.096)	(0.312)	(0.131)
<u>Early/mid seral stage by:</u>	0.307			
	(0.494)			
Mesozoic:		1.220 **		1.440 *
		(0.548)		(0.572)
granitic:			-1.130 *	-0.267
			(0.618)	(0.558)
sedimentary:		-0.404 *	-1.040 ***	
		(0.212)	(0.289)	
<u>Late seral stage by:</u>	0.6002 ***			
	(0.212)			
Mesozoic:		0.266		0.487
		(0.680)		(0.655)
granitic:			-0.577 *	0.291
			(0.340)	(0.193)
sedimentary:		-0.046	-0.694 *	
		(0.219)	(0.372)	
<u>Basin area by rock type</u>				
Mesozoic:	0.376			
	(0.383)			
granitic:	-0.265 ***			
	(0.088)			
sedimentary:	-0.307 *			
	(0.157)			
Latitude:	131.1 *	176.0 *	172.2 *	176.0 *
	(78.4)	(98.0)	(88.6)	(92.0)
<u>R-squared</u>	0.447	0.418	0.405	0.427

Notes: Standard deviations are in parenthesis. Significant t-tests are denoted as *=0.1 level; **=0.05 level; ***=0.01 level. R-squared values are adjusted to account for degrees of freedom; nevertheless they cannot be interpreted in the usual way due to the omission of the intercept. F tests indicate that all models are significant at the 0.001 level.

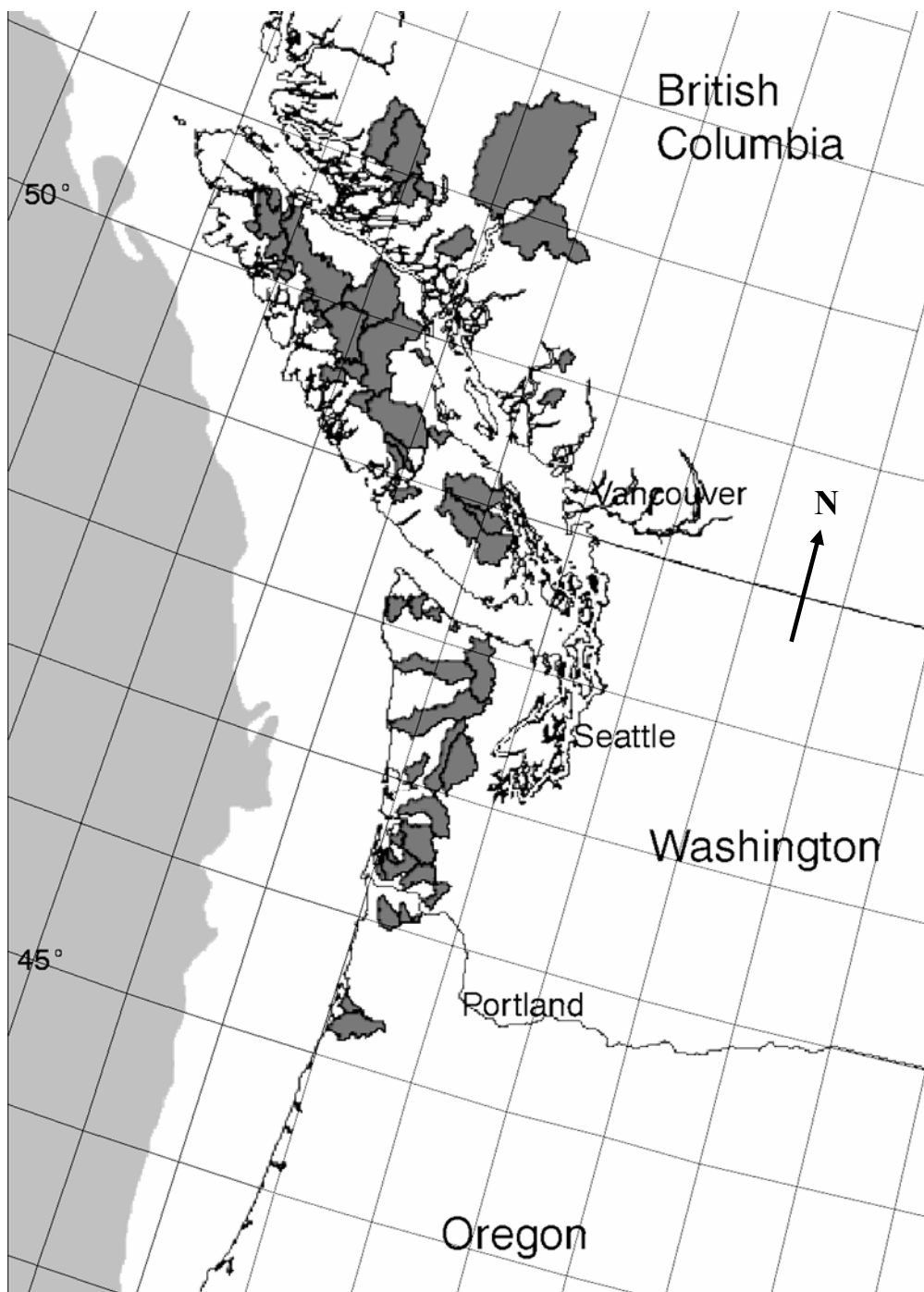


Figure 1. Location of the 62 watersheds in study sample.

(scale is 69.17 miles per degree latitude)