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HISTORICAL CHANGES IN POOL HABITATS IN THE COLUMBIA RIVER BASIN

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Abstract. An historical stream survey (1934–1945) was compared with current surveys (1987–1997) to assess changes in pool frequencies in the Columbia River Basin. We surveyed 2267 km of 122 streams across the basin, representing a wide range of lithologies, stream sizes, land use histories, ownerships, and ecoregions. Based on pool classes inherited from the historical surveys, the frequencies of large ($\geq$20 m^2 and $\geq$0.9 m depth) and deep ($\geq$20 m^2 and $\geq$1.8 m depth) pools have decreased significantly ($P < 0.01$) since the 1930s. We classified streams as natural or commodity based on their watershed management and land use histories. Natural streams were in watersheds minimally affected by human activities (e.g., wilderness or roadless designation, limited entry), with only 12% having roads in riparian areas. Commodity streams were defined as having watersheds managed predominantly for extraction of resources via timber harvest, livestock grazing, and other human activities. Ninety percent of these streams had roads in the riparian areas. In natural streams, large-pool frequencies increased or remained the same in 96% of the streams (88% for deep pools). In commodity streams, large- and deep-pool frequencies decreased in 52% and 54% of the streams, respectively. Despite differences in stream size and the level of human activities, the magnitude and direction of these changes were consistent. Land ownership did not influence trends; pools decreased significantly on both private and public lands. Only where entire watersheds or headwaters were designated as wilderness or roadless areas did pools consistently remain unchanged or increase. Pool frequencies decreased in all ecoregions except the North Cascades ecoregion. We developed regional histories of human activities for the Columbia River Basin. Human activity histories were typically of low spatial resolution and available for broad geographic areas only; we rarely were able to obtain information at the scale of individual watersheds. Consequently, we were unable to test the relationship between temporal and spatial patterns in human activities and their influence on site-specific trends in pools. Despite our inability to isolate causal mechanisms, management emphasis and human activities clearly influenced trends in pools. We conclude that the persistent effects of human activities have simplified stream channels and reduced large- and deep-pool frequencies in watersheds outside of designated wilderness and roadless areas in the Columbia River Basin.

Key words: aquatic restoration; Columbia River Basin; cumulative effects; decline of aquatic ecosystems; historical changes; habitat simplification; land use history; management emphasis; pool habitats.

Introduction

Pools are preferred habitats for many stream fishes during all or part of their freshwater life history (Elser 1968, Lewis 1969, Beschta and Platts 1986, Bisson et al. 1992). The habitat requirements of stream fishes vary depending on species, season, and life stage (Sullivan et al. 1987, Bisson et al. 1992). Pools provide rearing habitat for juvenile fish, resting habitat for adults (Bjornn and Reiser 1991), and refugia from natural disturbances, such as drought, fire, and ice (Sedell et al. 1990). During periods of thermal stress, fish use cool pools to behaviorally thermoregulate (Berman and Quinn 1991, Matthews et al. 1994, Nakamoto 1994, Nielsen et al. 1994, Torgersen et al. 1999). Pool–riffle interchange areas also provide important spawning sites (Reiser and Wesche 1977). Pools influence the diversity of stream fish communities (Bisson and Sedell 1984). As the volume and complexity of pools (i.e., diversity of cover, hydraulic, and substrate conditions) increase, the capacity to support a diversity of species and life stages also increases (Bjornn and Reiser 1991, Bisson et al. 1992, Fausch and Northcote 1992). Complex pools also produce larger fish biomasses (Fausch and Northcote 1992).

Knowledge of temporal changes in aquatic habitats within natural and human-influenced ecosystems is cur-

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rently limited at large regional scales and is largely based on anecdotal data such as trapper and emigrants' journals. An understanding of historical ecology is essential, as White and Walker (1997:347) have stated, "to learn how to link historical information and current conditions, determining the rules by which the past becomes the present and the present will lead to a range of possible future states." The most cited cause for the decline of aquatic ecosystems in the western United States is habitat loss due to land use practices (Williams et al. 1989, Nehlsen et al. 1991, U.S. Department of Agriculture 1993, Henjum et al. 1994). These summary conclusions have been based on paired-watershed (Hall et al. 1987), pre- and post-treatment (Lyons and Beschta 1983, Hartman and Scrivener 1990, Ryan and Grant 1991, Megahan et al. 1992), and space-for-time studies (Bisson and Sedell 1984, Bilby and Ward 1991, Overton et al. 1993, Reeves et al. 1993, Ralph et al. 1994) that evaluate the effects of one or a few activities on aquatic ecosystems. Although most of these studies suffer from limited scope (i.e., small spatial scales) and a lack of historical baselines and replicable measures for comparing streams (Ralph et al. 1994), the studies do provide information on the biophysical effects of specific activities on aquatic ecosystems.

The clearest examples of the accumulated effects of human activities on aquatic ecosystems are only evident in the most degraded rivers (Bisson et al. 1992). Examples include the South Fork Salmon River in Idaho (Megahan et al. 1992) and the Alsea River in Oregon (Hall et al. 1987). Human activities modify aquatic habitats by altering one or more of the following ecosystem attributes: channel structure, hydrology, sediment, water quality, riparian forests, and exogenous material (Gregory and Bisson 1997). The effects of logging, mining, livestock grazing, agriculture, or urbanization often are similar (Hicks et al. 1991) and result in simplification of stream channels and loss of habitat complexity (Bisson et al. 1992). Fish habitat simplification was defined by Reeves et al. (1993:309) as "a decrease in the range and variety of hydraulic conditions (Kaufmann 1987) and reductions in structural elements (Bisson et al. 1987), frequency of habitats, and diversity of substrates (Sullivan et al. 1987)." We will base references to fish habitat simplification in this document on the Reeves et al. (1993) definition.

Long-term monitoring data are essential for evaluating the effects of natural and anthropogenic disturbances on aquatic ecosystems (Sedell and Luchessa 1982). Sedell and others (Sedell and Luchessa 1982, Sedell and Froggatt 1984, Sedell and Duvall 1985, Sedell et al. 1988) pioneered the use of historical records to document the effects of Euro–American development on aquatic ecosystems in the Pacific Northwest. Recently, aerial photography has been used to quantify changes in aquatic ecosystems and relate them to natural and human-caused disturbances (Beschta 1983*a*, *b*, Lyons and Beschta 1983, Grant 1988, Ryan and Grant 1991, Smith 1993, Minear 1994).

In 1987, the U.S. Forest Service Pacific Northwest Research Station recovered a historical U.S. Department of Commerce Bureau of Fisheries stream habitat survey of the Columbia River Basin. These surveys were conducted during 1934–1945 to determine the condition of streams in the Columbia River Basin that provided, or had provided, spawning and rearing habitat for anadromous salmonids (*Oncorhynchus* spp.) (Rich 1948). Prior to our research, published summaries of the Bureau of Fisheries survey were largely documentation of the extent of potential migration barriers and irrigation diversions in the basin (Bryant 1949, Bryant and Parkhurst 1950, Parkhurst 1950*a*, *b*, *c*, Parkhurst et al. 1950). Our review of Rich (1948) indicated that the survey included the earliest and most comprehensive quantitative documentation of anadromous fish habitat in the Pacific Northwest. We were able to track the original survey data to a warehouse in Portland, Oregon, where they were destined for the incinerator. Unlike other historical surveys, the data were collected systematically, with replicable variables (e.g., pool and substrate classes) that allow a direct comparison to recent surveys (Bisson et al. 1992, McIntosh et al. 1994*a*, *b*, McIntosh 1995). The Bureau of Fisheries field data have allowed us to evaluate changes in pools in large basins, across a diverse region, with different land management histories, in a consistent, replicable manner.

Since 1987, we have been investigating streams originally surveyed by the Bureau of Fisheries. Our objectives were (1) to quantify trends in pool frequency and depth in the Columbia River Basin since the Bureau of Fisheries surveys; (2) to compare trends across streams of different management emphasis, ownership, and ecoregions; (3) to characterize, and quantify, where possible, the disturbance history in the basin; and (4) to identify potential causal mechanisms between disturbance history and trends in pools. Results from individual streams (Peets 1993, Smith 1993), large watersheds (McIntosh 1992, Minear 1994), and select regions of the Columbia River Basin (McIntosh et al. 1994*a*, *b*) have been published previously. This paper adds to the literature regarding the Bureau of Fisheries data by expanding the geographic scope of the research from individual streams and watersheds to a large catchment, the Columbia River basin. We also analyzed the spatial distribution of changes in pool habitats utilizing stream and watershed classification.

Materials and Methods

The Columbia River Basin encompasses parts of seven states and one Canadian province, with a drainage area of 667 000 ha. Based on discharge, it is the second largest river basin in the United States. Before Euro-American development, 23 617 km of streams in the Columbia River Basin were accessible to anadromous

TABLE 1. Pool size classes used in the historical (1934–1945) and current (1987–1997) stream habitat surveys.

Area/depth criteria	Historical pool class	Current pool class†
>40 m² area and >1.8 m depth	S1	large, deep
20–40 m² area and 0.9–1.8 m depth	S2	large
20–40 m² area and 0.7–0.9 m depth	S3	…
20–40 m² area and >1.8 m depth	S4	large, deep
>40 m² area and ≥0.7–0.9 m depth	S5	…
Small pools in cascades and behind boulders	S6	…

† We did not use S3, S5, and S6 pool classes in the comparisons. The S3 and S5 classes had narrow depth criteria (≥0.7 to 0.9 m depth), and S6 pools lacked objective criteria.

fish (Thompson 1976). Approximately 7396 km of these streams are no longer accessible to anadromous fish, and much of the remaining habitat has been degraded by human activities (Northwest Power Planning Council 1986).

Stream habitat surveys

The Bureau of Fisheries surveys were conducted during 1934–1945, inventorying 390 streams representing >6400 km of river in the Columbia River Basin. Habitats for spring chinook salmon (*Oncorhynchus tshawytscha*) were the primary focus of the surveys (Rich 1948). Rich (1948) provided a detailed description of methods used in the Bureau of Fisheries survey. Data were systematically sampled upstream in consecutive 91-m (100-yard) intervals for the entire section surveyed, typically during the summer, from the confluence upstream to the upper limit of anadromous fish use. Within each 91-m unit, the surveyors visually estimated mean channel width, substrate particle size composition by four classes, and the number of pools in six classes. Size–class criteria are shown in Table 1.

We resurveyed the same sections during 1987–1997 at summer low flow, using a stratified sampling technique (Hankin and Reeves 1988) based on aquatic habitat units such as pools, riffles, and glides, as defined by Bisson et al. (1982). In the current surveys, as opposed to the arbitrary length used in the Bureau of Fisheries survey, sampling units were defined by morphological characteristics, such as surface turbulence, water velocity, and channel cross section. To verify that our methods were consistent with the historical survey, we brought the last living member of the Bureau of Fisheries study, Professor David Frey of the University of Indiana, to Corvallis and interviewed him regarding the survey methodology. He brought extensive diaries, along with sharp memories, of his two years with the Bureau of Fisheries, which included the surveys of the Grande Ronde, Salmon, and Willamette basins. Frey verified the methods they used, leaving us confident that the data was replicable. We concluded that the Bureau of Fisheries pool classes S1–S4 meet the criteria for pools as defined by Bisson et al. (1982). These pools were the large, deep, low-velocity areas that provide resting and holding habitats for salmon. We compared Bureau of Fisheries pool counts and current counts to assess trends in pools in the Columbia River Basin. In conducting the current surveys, we attempted to examine streams across the Columbia River Basin encompassing as much variation in geologic conditions, land ownerships, and land use histories as possible (Fig. 1).

To reduce observer bias, we combined the Bureau of Fisheries pool classes into two categories for this study: (1) large pools (≥20 m² area and ≥0.9 m depth; all S1, S2, and S4 pools), and (2) deep pools (≥20 m² area and ≥1.8 m depth; all S1 and S4 pools). We did not use S3, S5, and S6 pool classes in the comparisons. The S3 and S5 classes had narrow depth criteria (0.7–0.9 m depth), and S6 pools lacked objective criteria. By eliminating these size classes, we reduced the potential for observer bias, because classification errors were less likely. We believe the possible surveyor bias in the original Bureau of Fisheries surveys was also reduced when we used the two broad size classes.

We also addressed potential observer bias between surveys in a second way. In the current survey, we calculated habitat areas (Hankin and Reeves 1988) and measured the maximum depth of each pool at low flow. To account for interannual hydrologic variability and potential observer bias, we imposed an intentional bias towards more pools in the current survey. Marginal pools were discarded in the Bureau of Fisheries survey, but were included in the current study. In the Bureau of Fisheries surveys, marginal pools were those surveyors had noted as shallow or small. Only large pools ≥0.9 m depth and deep pools ≥1.8 m depth were included in the historical survey. In the current survey, large pools ≥0.8 m depth and deep pools ≥1.6 m depth were included. In effect, the data represent a bias for fewer pools historically and more pools currently. We believe this approach provides a conservative estimate of changes in pool frequencies. The streams we used for comparison were not randomly or systematically selected from the Bureau of Fisheries data set. Instead, because of funding constraints or the availability of data, we were opportunistic in the streams we compared. Only 58% of the streams were surveyed specifically for this study. The remainder of the data was collected by other agencies and institutions using the Hankin–Reeves method as part of their stream habitat survey programs. We are confident that large and deep pools were consistently identified using this method. A two-sample *t* test (Zar 1996) was used to test whether our subsample of streams was representative of the entire Bureau of Fisheries data set.

By comparing the two surveys, we can assess the quantity and quality of fish habitat, both historically and currently. We used the frequency of large pools as

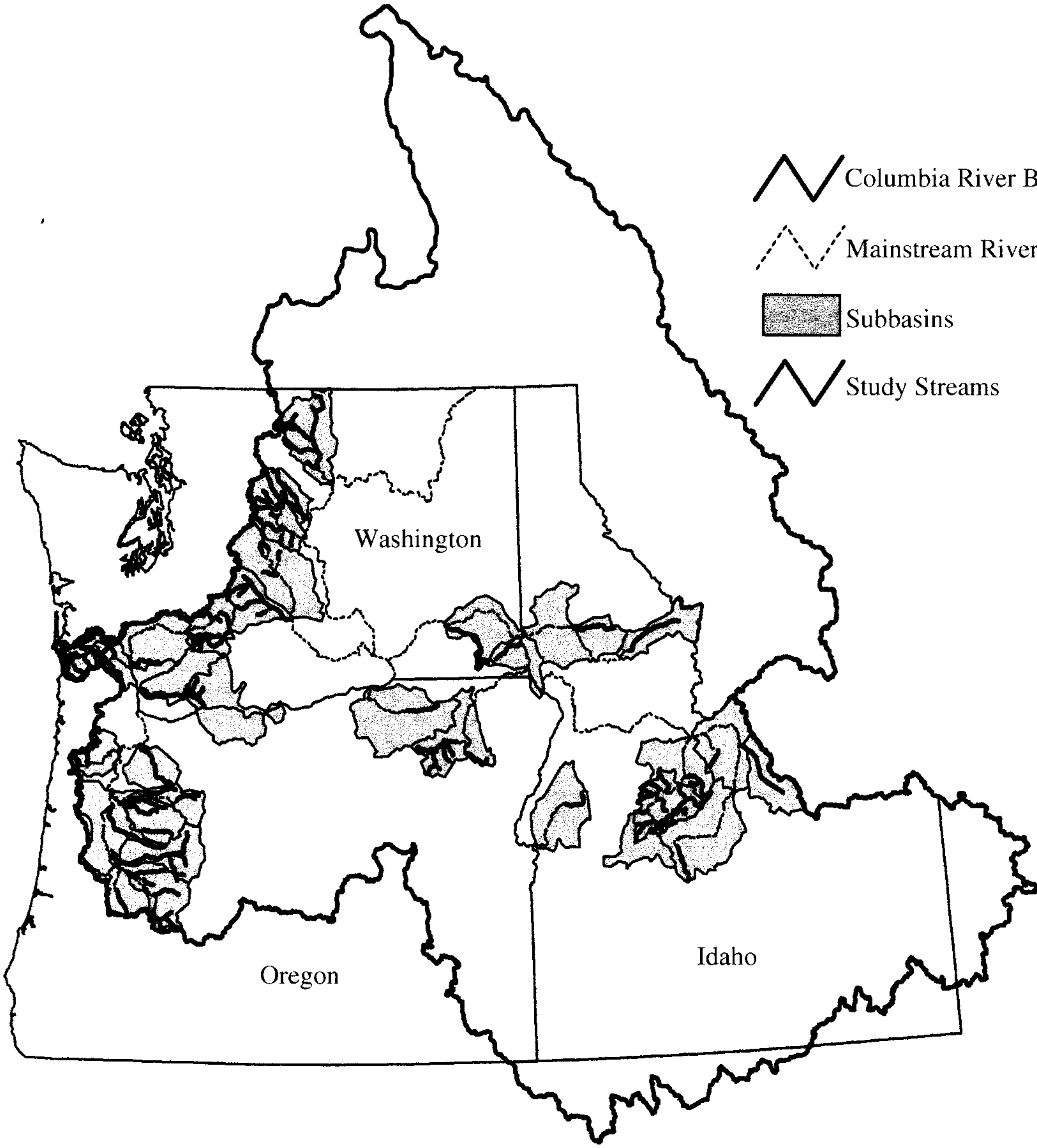

FIG. 1. Map of Columbia River Basin showing study basins and streams surveyed.

indicators of the quantity of pools, and used the frequency of deep pools as indicators of habitat quality. Factors that affect habitat quality include velocity, depth, substrate, temperature, and cover (Bjornn and Reiser 1991). Most pools in Pacific Northwest streams are formed around local obstructions (Sullivan et al. 1987), particularly in areas where large woody debris is the most abundant in-channel structural element (Montgomery et al. 1995). The deepest pools result from scour produced by strong secondary currents around bends and obstructions (Sullivan et al. 1987). Deeper pools contribute to higher quality habitat by providing refuge from terrestrial predators (Bisson et al. 1987) and summer low flows (Beschta and Platts 1986). Deeper pools may also increase fish community diversity by allowing fish species and age classes to segregate in the water column (Fraser 1969, Allee 1982).

The frequency at which pools occur is a fundamental aspect of fluvial geomorphology (Leopold et al. 1964). In free-formed pool–riffle reaches, pools tend to occur every 5–7 channel widths (Leopold et al. 1964, Keller and Melhorn 1978) and every 1–4 channel widths in steeper, step–pool reaches (Grant et al. 1990). Pools may be freely formed by the interaction of sediment and flow, or forced by local obstructions, such as large woody debris, bedrock, root masses, and debris jams, which cause local scour (Beschta and Platts 1986, Montgomery et al. 1995). Forced pool morphologies can increase the natural variability in pool spacing (Beschta and Platts 1986), often reducing the distance between pools (Montgomery et al. 1995).

We attempted to account for the effect of stream size on pool frequency by stratifying the study streams using several surrogates for stream size. These included drainage basin characteristics (drainage area, Strahler [1952]; stream order), hydrology (mean annual discharge), and channel characteristics (mean wetted channel width). Drainage area and stream orders were derived from 1:100 000-scale United States Geological Survey (USGS) topographic maps, and discharge data were obtained from USGS gauging stations. We calculated mean wetted channel widths from the Bureau of Fisheries survey to quantify channel width.

In order to analyze changes in large and deep pools, we standardized the pool data by calculating means (number of pools per kilometer surveyed) for each stream. These values were then used to calculate the grand mean (mean of the stream means), standard deviation (sd) of the grand mean, and the range for the two surveys. A paired two-sample *t* test (Zar 1996) was used to test for differences in pool frequencies between the surveys. We examined pool data for normality and transformed nonnormal data using the square root function (Zar 1996). We used 95% confidence intervals to determine if the net change in pool frequencies between the two surveys was statistically significant. Confidence intervals were calculated from the net change in pool frequencies between the two surveys and positioned around zero (no change). We concluded the sign of change (±) was statistically significant from zero (no change) when the net change in pool frequencies from the Bureau of Fisheries to the current surveys was greater than the 95% confidence interval.

Stream classification

To assess the spatial distribution of fish habitat changes and the influence of management, we classified each study stream according to management emphasis, land ownership, and ecoregions. For management emphasis, we classified each stream as natural or commodity based on the current and historical land use of the watershed. Natural streams were in watersheds that were minimally affected by human disturbance (e.g., wilderness or roadless designations, limited entry). Natural streams were not pristine; however, despite some historical human influences (e.g., mining, grazing) and a policy of fire suppression over the past century, streams in natural watersheds provide a relative baseline of natural change. Commodity streams were primarily in roaded watersheds managed predominantly for extraction of resources via timber harvest, livestock grazing, agriculture, and mining. Stream reaches were classified as private or public based on ownership of the reach, regardless of the management emphasis of the watershed. We recognize that downstream effects may have influenced the condition of individual reaches, but suggest this analysis is still useful for a comparison of changes in pool frequencies on public and private lands. A paired two-sample *t* test (Zar 1996) tested for differences in pool frequencies within management emphasis and ownership classes. We examined pool data for normality and transformed nonnormal data using the square root function (Zar 1996).

To assess regional patterns in pools, we stratified data by ecoregions. Ecoregions are based on regional differences in landforms, potential natural vegetation, soils, and land use (Omernik and Gallant 1986, U.S. Environmental Protection Agency 1996). Hughes et al. (1987) and Whittier et al. (1988) found significant relationships between ecoregions and spatial patterns in stream ecosystems in the Pacific Northwest. Our study streams were located in the Coast Range, Western Cascades, North Cascades, Blue Mountains, and Northern Rockies ecoregions. Characteristics of these ecoregions are described in detail in by Omernik and Gallant (1986) and the U.S. Environmental Protection Agency (1996). We used a one-way ANOVA and, if significant differences ($\alpha < 0.05$) were found, a Tukey test for unequal sample sizes (Zar 1996) to detect differences between ecoregions. Nonnormal data were transformed by using square root functions.

Land use history

The Atlas of the Pacific Northwest (Jackson and Kimerling 1993) provides a general overview of land use in the Pacific Northwest. Commercial timberland (both public and private) is the dominant land use in the region (35.6% of the land base), followed by grazing lands (35.2%), croplands (15.9%), noncommercial timberlands (10.2%), developed lands (urban and transportation, 1.8%), and National Parks (1.3%). Based on the this data, forestry and livestock grazing are the dominant land uses in the region by area (70.8%). As the dominant land uses in the region, we focused on quantifying the magnitude and extent of these activities since the beginning of Euro-American settlement. We developed quantitative timber harvest and grazing histories for the five ecoregions encompassing our study streams, and we examined these records to characterize their magnitude and extent before and after the Bureau of Fisheries survey. While other land uses, such as agriculture, mining, and urbanization, have clearly affected streams in the region, we submit that the effects of forestry and livestock grazing practices on fish habitat simplification are widespread and pervasive. As Gregory and Bisson (1997) noted, the effects of livestock grazing on aquatic ecosystems have been consistent with observed responses on forested lands.

Livestock numbers for Idaho, Oregon, and Washington were derived from United States census data by counties at 10-yr intervals during 1850–1950 (U.S. Department of Commerce 1910–1950, U.S. Department of the Interior 1850–1900) and at five-year intervals during 1954–1992 (U.S. Department of Commerce 1954–1992). Livestock data consisted of the population counts by livestock class. We analyzed trends in beef cattle (excluding dairy cows) and sheep, as these are the dominant livestock in the region. Dairy cows were excluded because their range is typically confined to pastures and feedlots. Most native rangelands are utilized by beef cattle and sheep. These contrasting management approaches are likely to result in local effects (dairy cows) vs. watershed effects (beef cattle) on stream habitats. The volume of timber harvested was available for Oregon and Washington on a statewide basis during 1869–1924 (Oregon Department of Forestry 1943; D. Larsen *unpublished data*) and on a countywide basis during 1925–1994 (Oregon Department of Forestry 1951–1994, Washington State Department of Natural Resources 1951–1994, Wall 1972). In Idaho, annual harvest volumes during 1948–1993 were available only for the Nez Perce and Clearwater National Forests.

RESULTS

We analyzed 2267 km of 122 streams in 16 river basins for changes in pools (Fig. 1). This represents 31% of the streams and 35% of the stream lengths surveyed historically, with streams from 16 of the 21 surveyed river basins. Stream segments ranged from 0.8–122.1 km (mean, 18.6 km; sd, 19.4 km), and stream size ranged from small headwater streams (drainage area <50 km^2) to large rivers (drainage area >4700 km^2). We analyzed data from across the Columbia River Basin representing a broad range of stream types and human disturbance histories. We found no significant differences ($P > 0.05$) in large-pool frequencies, but significant differences ($P < 0.05$) in deep-pool frequencies between our subsample of historical streams and the entire historical database. Therefore, our subsample of streams was representative of the frequency of large pools, but deep-pool frequencies were slightly higher in the subsample than in the complete data set.

We found no significant relationships between pool frequency and any measures of stream size for the historical or current data set. This analysis may be incomplete, however, because of inadequate measures of stream size, availability of data, or a poor understanding of the processes that determine pool formation across such a wide range of stream sizes. For example, Montgomery and Buffington (1993) concluded that stream order was a useful tool for describing channels within a watershed, but inadequate for comparing watersheds because of differences in drainage densities between watersheds and inconsistencies in mapping of stream channels. We suspect measures of bankfull-width and channel gradient might be more appropriate, but these measurements were either unavailable or beyond the scope of this study.

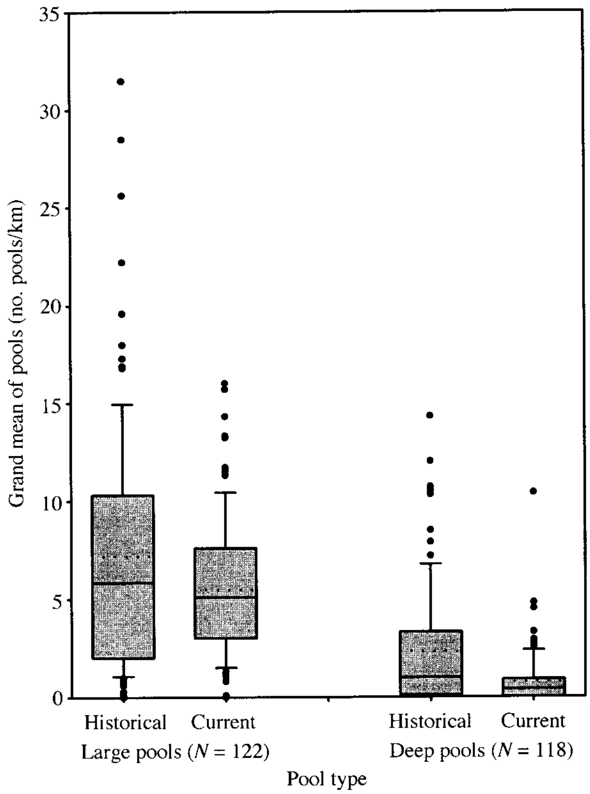

FIG. 2. Frequency of large pools and deep pools in the historical and current surveys. In the box plots, dotted and solid lines denote grand mean and median, respectively; boxes show 25th and 75th percentiles; whiskers show 10th and 90th percentiles; and circles denote outliers.

Changes in pools

The frequency of large and deep pools in the Columbia River Basin decreased significantly (paired two-sample t test, $P < 0.01$) from 1934–1945 to 1987–1997. Large pools decreased by 24% (from grand means of 7.2 to 5.5 pools/km, $N = 122$ streams; Fig. 2) and deep pools by 65% (from grand means of 2.3 to 0.8 pools/km, $N = 118$ streams; Fig. 2). The variance and range in pool frequencies also decreased from the Bureau of Fisheries surveys to the current surveys. Deep pools were 36% of the pools counted in the historical survey and decreased to 17% of the pools in the current survey.

Changes in pools based on management emphasis

Management emphasis class influenced the trend in large and deep pools between the two surveys. We classified 25 streams as natural and 113 as commodity, based on management emphasis and land use history. Twelve of the study streams were not used for deep pool comparisons because of incomplete data. Stream orders were similar for natural and commodity streams, with third- and fourth-order streams being the most common. Natural streams ranged from third- to fifth-order (median, 4; sd, 0.5) and commodity streams

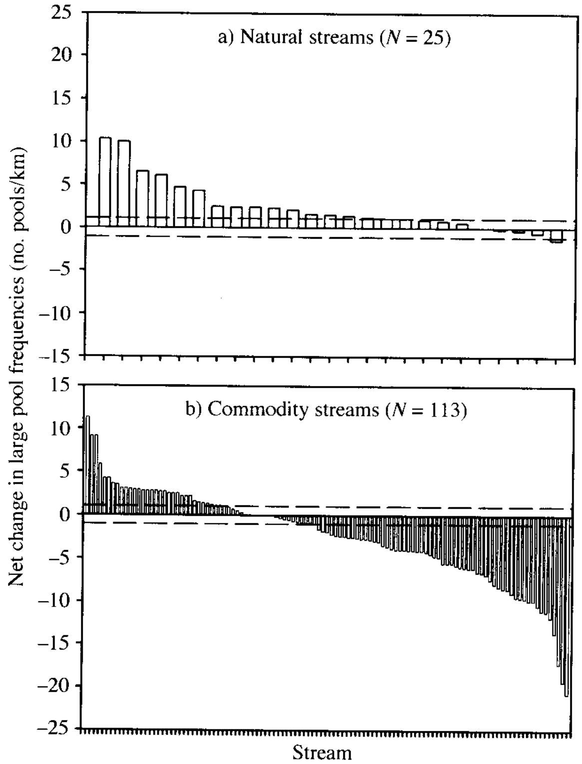

FIG. 3. Net change in large-pool frequencies in (a) natural and (b) commodity streams, from the historical to current surveys. Bars represent the net change in pool frequency for each stream resurveyed, and the dotted line is the 95% confidence interval for the sign of change (±) from zero (no change).

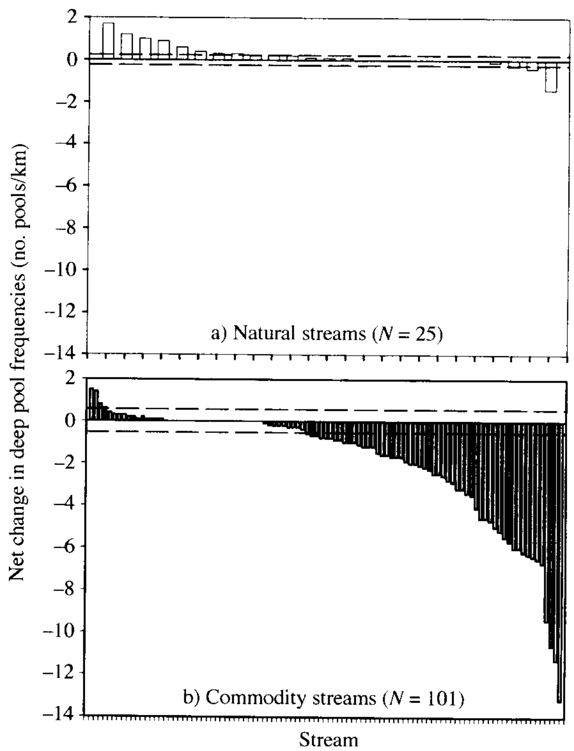

FIG. 4. Net change in deep-pool frequencies in (a) natural, and (b) commodity streams, from the historical to current surveys. Bars represent the net change in pool frequency for each stream resurveyed, and the dotted line is the 95% confidence interval for the sign of change (±) from zero (no change).

ranged from second- to sixth-order (median, 4; sd, 0.9). Large-pool frequencies increased by 81% in natural streams (from grand mean of 3.1 to 5.6 pools/km, N = 25, $P < 0.01$) and decreased by 32% in commodity streams (from grand mean of 7.9 to 5.4 pools/km, N = 113, $P < 0.01$). In natural streams, large-pool frequencies increased or remained unchanged in 96% of the streams surveyed and decreased in only 4% of the study streams (Fig. 3a). Large-pool frequencies in commodity streams decreased in 52% of the streams surveyed and increased or remained unchanged in the remaining 48% (Fig. 3b).

Deep-pool frequencies increased by 50% in natural streams (from grand mean of 0.4 to 0.6 pools/km, N = 25, $P < 0.05$) and decreased by 71% in commodity streams (from grand mean of 2.7 to 0.8 pools/km, N = 101, $P < 0.01$). The frequency of deep pools increased or remained unchanged in 88% of the natural streams surveyed and decreased in 12% (Fig. 4a). Deep pools decreased in 54% of the commodity streams surveyed and increased or remained unchanged in 46% (Fig. 4b).

Our intentional bias towards more pools in the current survey provides a conservative estimate of decreases in pool frequencies, but overestimates increases in pool frequencies. Natural streams were the only streams classified by management emphasis that showed increases in pool frequencies. We reanalyzed these data sets with the bias towards more pools in the current survey removed in order to test whether the increases we reported were still statistically significant. Our results indicate that the increases in large-pool frequencies in natural streams were reduced slightly (Table 2), but were still highly significant ($P < 0.01$).

TABLE 2. Analysis of changes in large- and deep-pool frequencies in natural streams with and without pool depth bias in current stream habitat surveys.

Pool class	Historical survey (no. pools/km)	Current survey (no. pools/km)	P
Large pools			
Maximum depth (≥0.8 m)	3.1	5.6	<0.01
Maximum depth (≥0.9 m)	3.1	4.9	<0.01
Deep pools			
Maximum depth (≥1.6 m)	0.4	0.6	<0.05
Maximum depth (≥1.8 m)	0.4	0.5	0.21

Note: A paired two-sample t test (Zar 1996) was used to test for differences between the surveys.

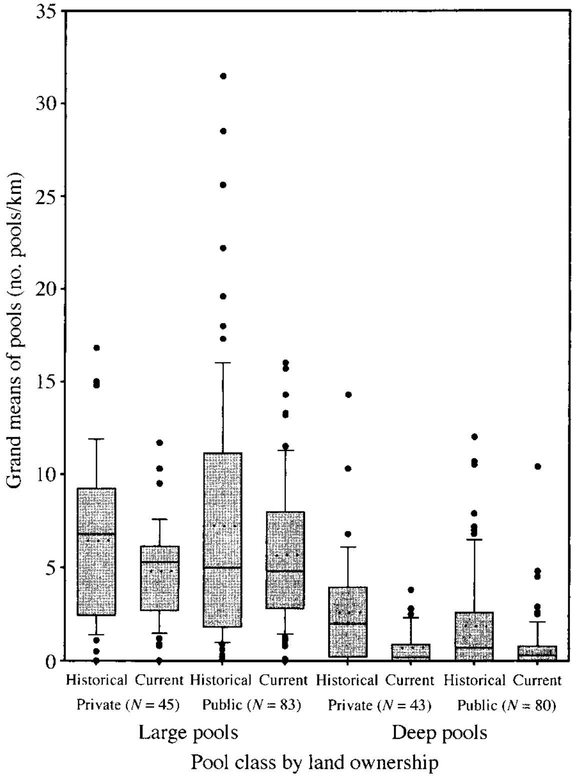

FIG. 5. Frequency of large and deep pools on private and public lands in the historical and current surveys. In the box plots, dotted and solid lines denote median and grand mean, respectively; boxes show 25th and 75th percentiles; whiskers show 10th and 90th percentiles; and circles denote outliers.

Increases in deep pools in natural streams were not significant ($P = 0.21$, Table 2) in the reanalyzed data, indicating that there were no differences in deep-pool frequencies between the two surveys.

When the historical survey was conducted, large- and deep-pool frequencies were significantly lower in natural streams than in commodity streams (two-sample t test, $P < 0.05$). In the current surveys there were no significant differences in large-pool frequencies (two-sample t test, $P = 0.86$) and deep-pool frequencies (two-sample t test, $P = 0.25$) between natural and commodity streams. From the historical to the current survey, the variance and range about the mean remained unchanged in natural streams and decreased in commodity streams.

Changes in pools based on land ownership

The frequency of large and deep pools decreased significantly ($P < 0.01$) on both private and public lands from 1934–1945 to 1987–1997. There were no significant differences in pool frequencies between private and public lands for either the historical or current survey. Stream orders ranged from second to sixth on private lands (median, 4; sd = 1.1) and second to fifth on public lands (median, 4; sd, 0.7), with third- and fourth-order streams being the most common for both ownerships. Large-pool frequencies decreased by 25% on private lands (from grand mean of 6.4 to 4.8 pools/km, $N = 45$, $P < 0.01$) and 21% on public lands (from grand mean of 7.3 to 5.7 pools/km, $N = 83$, $P < 0.01$; Fig. 5). Deep-pool frequencies decreased by 73% on private lands (from grand mean of 2.6 to 0.7 pools/km, $N = 43$, $P < 0.01$), while decreasing by 58% on public lands (from grand means of 1.9 to 0.8 pools/km, $N = 80$, $P < 0.01$; Fig. 5). The variance and range in pool frequencies was greater in the Bureau of Fisheries surveys than in the current surveys.

Changes in pools based on ecoregions classification

Changes in large and deep pools were significantly different ($P < 0.01$) between ecoregions. Large-pool frequencies increased significantly ($P < 0.01$) in the North Cascades ecoregion, while there were significant decreases ($P < 0.01$) in the Western Cascades, Blue Mountains, and Northern Rockies ecoregions (Fig. 6a).

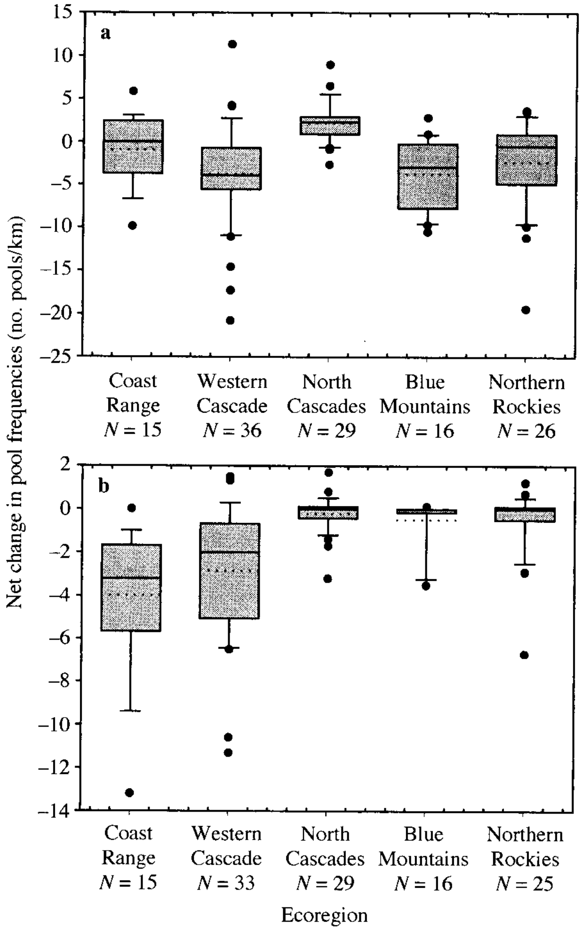

FIG. 6. Net change in the frequency of (a) large and (b) deep pools, by ecoregion. In the box plots, solid and dotted lines denote median and grand mean, respectively; boxes show 25th and 75th percentiles; whiskers show 10th and 90th percentiles; and circles denote outliers.

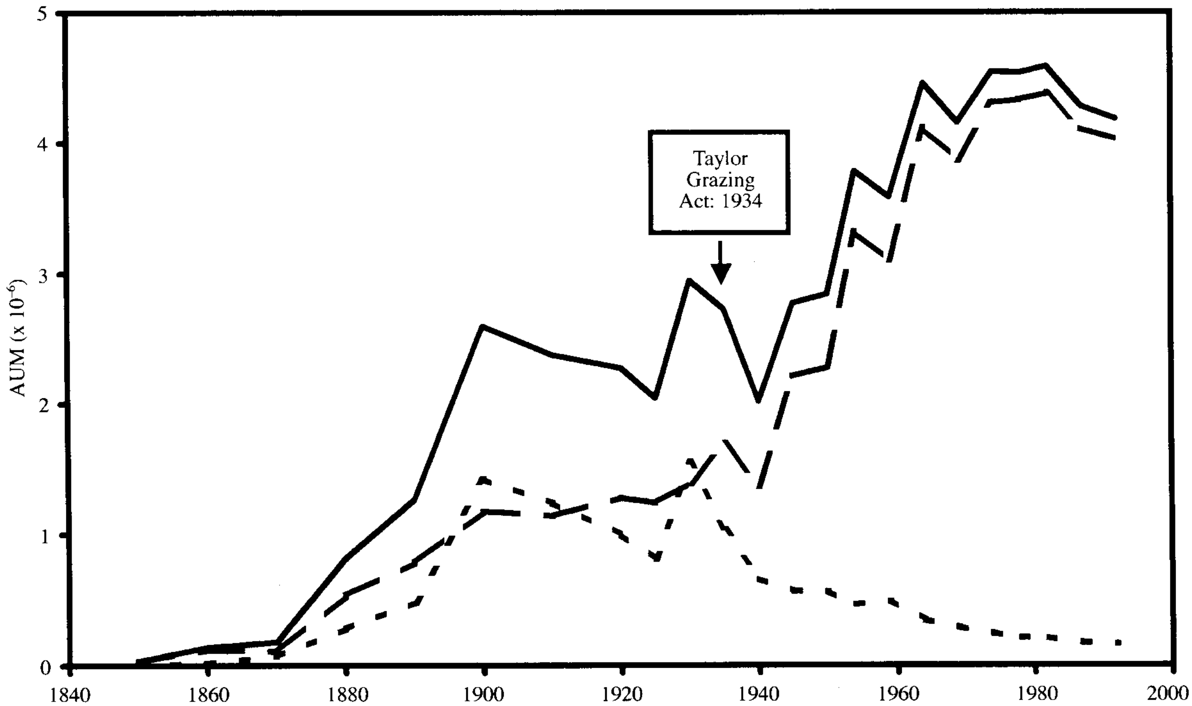

FIG. 7. Livestock use in animal unit months (AUM) in the Pacific Northwest during 1850–1992. Five sheep equal one cow for AUM calculations. The arrow indicates the timing of the Grazing Act. Solid line, total livestock use; long dashes, cattle use; short dashes, sheep use.

There were no differences between the Coast Range ecoregions and the other regions. The Coast Range and Western Cascades ecoregions had larger decreases in deep pools than the North Cascades, Blue Mountains, and Northern Rockies ecoregions (Fig. 6b). Because natural streams were rare in some ecoregions, we conducted a second analysis involving only commodity streams. The results remained highly significant ($P < 0.01$), and the regional differences in large and deep pools did not differ from the earlier analysis. Pools in the North Cascades ecoregion increased in both management emphasis types, although the increase was nearly twice as large in natural streams as in commodity streams. Of 22 streams surveyed in the North Cascades ecoregion, only one showed a decrease in large pools and six in deep pools.

Disturbance history

Quantitative records of land use practices in the Columbia River Basin were not available on a river basin-specific basis, but were generally available by counties or National Forests. For more qualitative reviews of the land use history of the Pacific Northwest, see Sedell and Duvall (1985), Northwest Power Planning Council (1986), McIntosh et al. (1994*a, b*), Robbins and Wolf (1994), Wissmar et al. (1994*a, b*), McIntosh (1995), Spence et al. (1996), and Dwire et al. (1999). We used data that corresponded closely to the study basins and further aggregated these data to the scale of Aquatic ecoregions.

Livestock grazing.—The first cattle were brought to the Pacific Northwest in 1789 by Spaniards, to Vancouver Island, British Columbia, Canada (Galbraith and Anderson 1991). In 1825, cattle were found in isolated settlements on Puget Sound and the lower Columbia River. By 1850, livestock were common in the scattered Euro-American settlements throughout the region. These early settlements were concentrated in the Willamette Valley and lower Columbia River and expanded inland with the discovery of gold in the 1860s (Robbins and Wolf 1994). Settlers soon recognized the economic potential for livestock, as well as the seemingly abundant forage and water resources to support them. As immigration increased on the Oregon Trail in the mid-1800s, the livestock industry grew rapidly in the Pacific Northwest (Robbins and Wolf 1994).

Analysis of United States census data shows that livestock use steadily increased in the Pacific Northwest during the period 1850–1992 (Fig. 7). Population numbers were converted to animal unit months (AUM) to standardize the relative effect of different livestock classes. For this analysis, five sheep were equivalent to one cow (Heady 1975). From 1850–1900, livestock increased from 35 000 to 2.6 million AUMs in response to growing Euro-American populations. By 1900, livestock was found throughout most of the Pacific Northwest (Robbins and Wolf 1994). From 1850–1905, grazing on Pacific Northwest rangelands was not regulated. There was intense competition among users, with no control on the number of animals or the season of use (Stoddart et al. 1975). The lack of any grazing management led to severe range damage. An 1883 report (Gordon et al. 1883) noted the almost immediate effect of this large-scale introduction of livestock. Overgrazing had damaged rangelands throughout the interior Columbia Basin (Gordon et al. 1883). This was the era of the open range, where there was no concept of land ownership or grazing management. While cattle ranchers fought among themselves for preferred range, most of their anger was directed at sheep ranchers (Galbraith and Anderson 1991). Cattle ranchers believed that the millions of sheep hooves destroyed the grass and the smell of sheep was a deterrent to cattle and horses using the range.

Despite the concern of ranchers and government officials, livestock grazing continued to increase until the turn of the century when Forest Reserves (predecessors of today's National Forests) were established. An 1898 National Academy of Sciences report focused national attention on overgrazing in the Forest Reserves (Irwin et al. 1994). Active management of grazing on Forest Reserves began in 1905 with the formation of the United States Forest Service. As public oversight of livestock grazing increased, livestock AUMs decreased by 23% from 1900–1925. In the period 1925–1930, however, livestock grazing increased by 45% to an historical high of 2.9×10^6 AUMs. Renewed public concern resulted in the passage of the Taylor Grazing Act in 1934, bringing grazing on the remainder of public lands under Federal management. The intent of the Taylor Grazing Act was to "stop injury to the public grazing lands by preventing overgrazing and soil deterioration, to provide for the orderly use, improvement and development [of public grazing lands], and to stabilize the livestock industry dependent upon the public range" (National Research Council 1994). The Grazing Service (which later became the Bureau of Land Management) was formed to create grazing districts and manage the public rangelands. Subsequently, livestock AUMs decreased 31% during 1935–1940. The decline in livestock AUMs was short lived, however; AUMs more than doubled during 1940–1992, increasing from 2.0 to 4.2×10^6 AUMs.

In addition to the virtually continuous increase in livestock grazing since 1850, the type of livestock grazing in the Pacific Northwest has changed from 1850 to 1992. Before 1920, cattle and sheep AUMs were approximately equal, but in 1920 cattle AUMs surpassed sheep AUMs; cattle currently account for 96% of the AUMs in the Pacific Northwest.

Timber Harvest.—Timber harvest in the Columbia River Basin began as Euro-American settlers migrated over the Oregon Trail in the mid-1800s. The first commercial sawmills were built at the mouth of the Columbia in Oregon in 1844 (Farnell 1981). With the California gold rush of the mid-1800s came an increased demand for timber, and log shipments from western Oregon supplied a large portion of the increased demand (Robbins and Wolf 1994). The earliest harvests were next to major rivers and streams, where the waterways were used as log highways (Sedell et al. 1991). By the 1880s, timber had been cleared along most major rivers and streams in western Washington and Oregon (Sedell and Luchessa 1982).

In eastern Oregon and Washington, along with Idaho, timber harvest started later, in response to the gold rushes in the interior Columbia Basin in the early 1860s. Lumber mills were built to support the mines and local markets (Robbins and Wolf 1994). Between 1860 and 1880, timber near the mining districts was sufficient to meet local demands. The national timber industry changed rapidly during 1880–1900 as the industry moved from the Great Lakes region to the Pacific Northwest and the railroads arrived in the region (Robbins and Wolf 1994). By the beginning of the 20th century, the timber industry in the Pacific Northwest was supplying both local and national needs (Robbins and Wolf 1994).

The systematic collection of timber harvest volume records for all land ownerships in the Pacific Northwest began in 1869. These records were available for Oregon and Washington, but not Idaho. Harvest volume in Oregon and Washington grew rapidly throughout 1869–1929, when it reached 29.7×10^6 m^3/yr (Fig. 8). The Great Depression temporarily depressed timber harvest after 1929, with the harvest volume in 1932 at its lowest level since 1908. It was not until 1941 that volume returned to a predepression level. After 1941, harvest volume grew steadily until 1968, when it peaked at 43.2×10^6 m^3/yr. During the period 1968–1989, harvest volumes cycled with regional and national economies. Since 1989, the listing of the Northern spotted owl (*Strix occidentalis caurina*) and anadromous salmonids under the Endangered Species Act has significantly curtailed timber harvest in the region (U.S. Department of Agriculture 1993, Quigley and Arbelbide 1997), with annual harvest volumes now at levels common prior to the 1920s.

We also determined regional patterns of timber harvest based on ecoregions. Our analysis by ecoregions shows harvests started in the Coast Range ecoregion and proceeded inland. In the Coast Range ecoregion, harvest had peaked by 1925 and has gradually decreased since then (Fig. 9). Timber harvest in the Western Cascades ecoregion was $<3.5 \times 10^6$ m^3/yr during 1925–1932, when it reached the recorded low of 0.9×10^6 m^3/yr. From 1932–1942, harvest volumes grew rapidly. Between 1942 and 1988, harvest fluctuated $5.9–10.1 \times 10^6$ m^3/yr until the listing of the Northern spotted owl under the Endangered Species Act began to limit timber supplies in 1989. In the North Cascades, Blue Mountains, and Northern Rockies ecoregions,

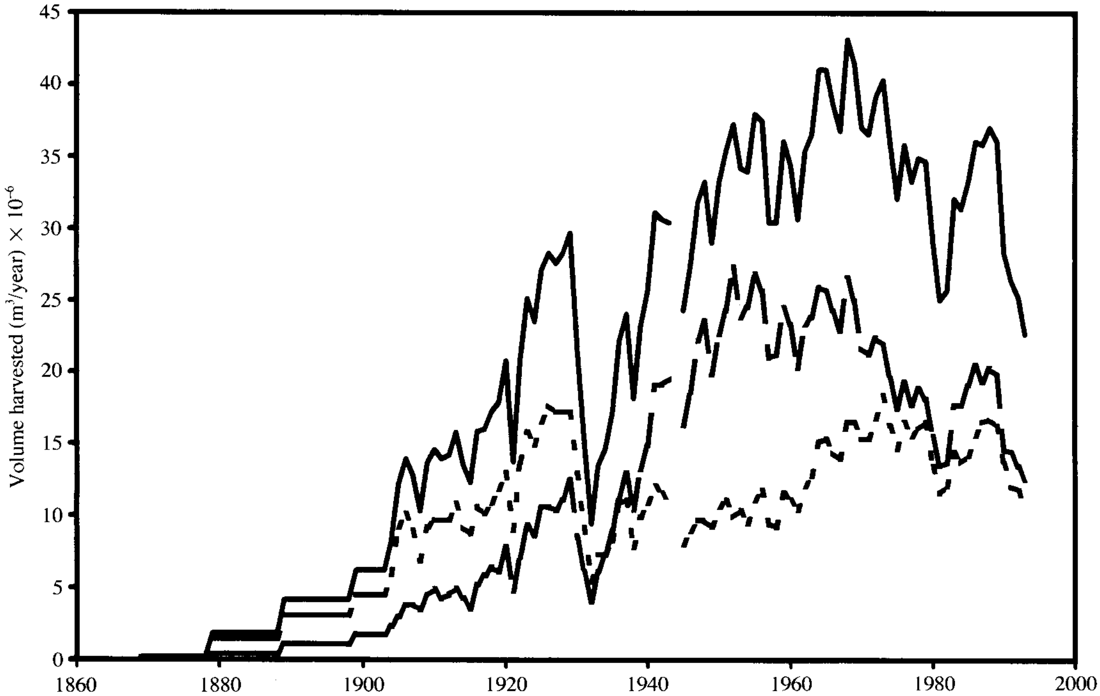

FIG. 8. Volume of timber harvested (millions of cubic meters per year) in Oregon and Washington during 1869–1994. Solid line, total; long dashes, Oregon; short dashes, Washington.

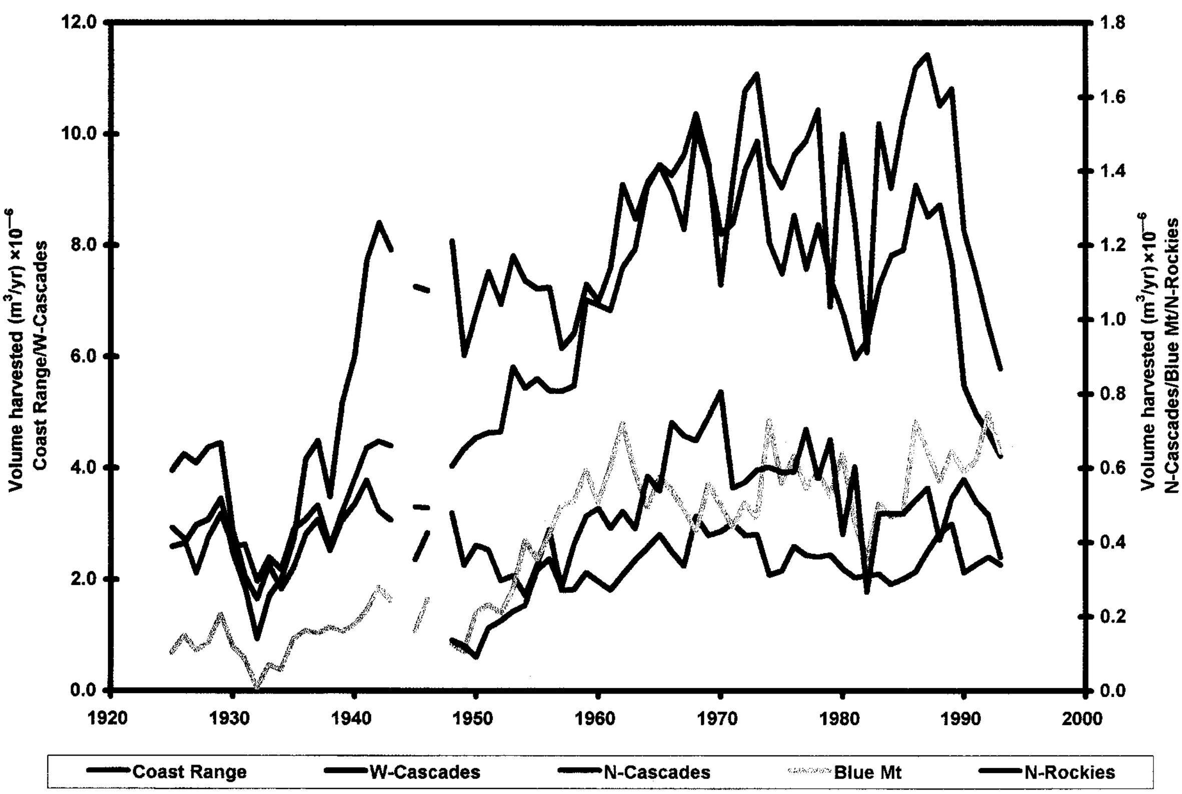

FIG. 9. Volume of timber harvested (millions of cubic meters per year) for study basins, by ecoregion, during 1925–1993.

timber harvest increased gradually from depression era lows until the 1960s and 1970s when harvest peaked. The listing of salmonids under the Endangered Species Act in the 1990s has reduced timber harvest in these regions.

After World War II, with the surplus of heavy machinery and availability of trucks (Oliver et al. 1994), roads became the dominant method for moving timber to the mills. Logging operations were no longer limited by lack of access to remote timber stands. Several recent reports have quantified the extent of the road network on public lands in the Pacific Northwest. The U.S. Department of Agriculture (1993) estimated that there were >175 000 km of roads and 250 000 stream crossings (culverts) within the range of the Northern spotted owl. This analysis extended from the Canadian border to just north of San Francisco, and from the Pacific Ocean to the east slopes of the Cascade range, an area of $\sim 10^7$ ha. From the Forest Ecosystem Management Assessment Team (FEMAT) report (U.S. Department of Agriculture 1993) and the Interior Columbia Basin Ecosystem Management Project (Quigley and Arbelbide 1997), we estimated the length of roads on public lands in the Columbia River Basin as >277 000 km. For our study streams, we found that 12% of the natural streams and 90% of the commodity streams had roads in adjacent riparian areas at the time of the current surveys.

Discussion

Our comparison of historical and current surveys demonstrates that the quantity and quality of pools in the Columbia River Basin have decreased significantly since the 1930s. By biasing our analysis in favor of more pools in the current survey and fewer in the historical survey, our work represents a conservative estimate of the magnitude of pool loss. We found a strong relationship between the management emphasis and land use history of the study watersheds and the direction of the change. The quantity and quality of pools increased or remained unchanged in natural streams, but decreased in commodity streams. The variability and range in pool frequencies remained unchanged in natural streams, but decreased among commodity streams over the study period. Previously published research using the Bureau of Fisheries data for individual streams (Peets 1993, Smith 1993), large watersheds (McIntosh 1992, Minear 1994), and select regions of the Columbia River Basin (McIntosh et al. 1994*a, b*) corroborate our results. The widespread and pervasive loss of pool habitats in commodity streams is not a local phenomenon, but a system-wide effect.

Historically deep pools were approximately seven times more frequent in commodity streams than natural streams; currently there is no difference in deep-pool frequencies between the two management emphases. The loss of deep-pool habitats has important implications for the biodiversity and productivity of native fish communities. Numerous researchers (Gerking 1949, Baltz and Moyle 1984, Bisson et al. 1992, Reeves et al. 1993, Matthews 1998) have shown that decreased pool depth reduces the availability of microhabitats, leading to decreased biodiversity of stream fishes. The effect is synergistic, as deep pools are typically associated with large woody debris (Abbe and Montgomery 1996). The ecological importance of large woody debris (Harmon et al. 1986, National Research Council 1992) and its affect on the distribution and abundance of salmonids in streams (Bisson et al. 1988) have been recognized in recent decades.

Decreases in pools were similar on private lands and public lands. This may indicate that the impacts of human activities are comparable among ownerships, or that lower elevation private lands were already strongly affected by human activities at the time of the Bureau of Fisheries survey, leading to a smaller magnitude of change between the historical and current surveys. Human activities, such as livestock grazing, timber harvest, splash dams, log drives, and mining, had already damaged lower elevation streams by the Bureau of Fisheries survey (Gregory and Bisson 1997), thus supporting the latter conclusion.

Ecoregion classification indicated that there was a regional pattern to this change, with all ecoregions except the North Cascades showing significant decreases in pools. These differences may due less to inherent differences in the resilience of ecoregions to human activities than to contrasting management emphases and land use histories. Management emphasis in the North Cascades ecoregion is predominantly wilderness, and human activities are more recent and have been at a lower magnitude than in the other ecoregions. It appears that management emphasis and land use history override the influence of ecoregions and ownership in determining the magnitude and direction of change in pools. McIntosh et al. (1994*a, b*) found a similar pattern when comparing central Washington streams to streams in northeast Oregon. In addition, deep pools had the largest decreases in ecoregions west of the Cascade Mountains. Current research in this region indicates that logjams form the deepest pools (Abbe and Montgomery 1996). The widespread removal of large woody debris in streams from the 1950s–1980s (Sedell et al. 1984), and reduced recruitment of large woody debris due to harvest practices, most likely led to a direct decrease in pool depth. Our results support the conclusion that a wide range of human activities causes the simplification (i.e., decreased quality and quantity) of stream habitats.

Previous research based on the Bureau of Fisheries data has shown similar trends at scales ranging from individual streams (Peets 1993, Smith 1993), through large watersheds (McIntosh 1992, Minear 1994), to select regions of the Columbia River Basin (McIntosh et al. 1994*a, b*). Research based on paired watersheds and pre- and post-treatment comparisons have shown that human activities simplify stream habitats. The results are similar, whether the dominant land use was grazing (Platts 1991), timber harvest (Bisson and Sedell 1984, Hartman and Scrivener 1990, Fausch and Northcote 1992, Frissell 1992, Megahan et al. 1992, Overton et al. 1993, Reeves et al. 1993, Ralph et al. 1994), mining (Nelson et al. 1991), agriculture (Karr and Schlosser 1978, Karr et al. 1983), flood control (Cederholm and Koski 1977, Chapman and Knudsen 1980), urbanization (Leidy 1984, Leidy and Fiedler 1985), or multiple use (Beechie et al. 1994).

Most research on the simplification of stream habitats has focused on the effects of timber harvest. Many researchers have found that pool frequency or area decreases significantly in logged watersheds, in response to lower levels of large woody debris (Hicks 1990, Bilby and Ward 1991, Overton et al. 1993, Reeves et al. 1993, Ralph et al. 1994, Montgomery et al. 1995), or in response to increased sediment delivery coupled with decreased large woody debris (Burns 1972, Frissell 1992). Increased sediment delivery and/or loss of pool-forming elements can result in decreased pool frequencies, regardless of the land use practices that caused it. Human activities can result in decreased pool frequencies by increased sedimentation (Lisle 1982, Megahan 1982, Jackson and Beschta 1984, Alexander and Hansen 1986, Lisle and Hilton 1992), or by the elimination of pool-forming elements, such as riparian vegetation, large woody debris, and boulders (Bilby 1984, Bisson and Sedell 1984, Sullivan et al. 1987, Hicks 1990, Fausch and Northcote 1992, Ralph et al. 1994). Alternatively, both processes may work synergistically. As Ralph et al. (1994) concluded, the biophysical effects of land use on streams are moderately well understood, but their extent and significance across broad regional landscapes are poorly documented. Our research provides clear documentation of the widespread and pervasive effects of land use on aquatic habitats from the scale of individual streams to a 667 000-ha basin.

The elimination of pools is the result of loss of riparian vegetation and pool-forming elements and increased sedimentation. Recovery of large, deep pools is likely to take decades, depending on the mechanisms of pool formation. For example, where large woody debris is the primary mechanism of pool formation, it may take centuries for trees to grow large enough to be recruited into streams. In ecosystems dominated by wet meadows and woody vegetation (e.g., willows and alders), the time span may be much less. The restoration of pool habitats will take even longer to recover if management activities that forestall recovery remain the status quo (e.g., road construction, riparian timber harvest, and livestock grazing leading to degraded riparian vegetation, low recruitment of large woody debris, and increased sediment delivery).

If pools have decreased in commodity streams because of human activities, what are the mechanisms for increased pool formation in natural streams? We offer the following hypothesis. Stochastic events, such as a fire followed by a large flood, can result in the episodic delivery of large woody debris and boulders to the stream channel (Reeves et al. 1995), increasing the potential for pool formation over time. If we assume that natural streams are high-integrity aquatic ecosystems, then floods are likely to cause major inputs of pool-forming elements, such as large woody debris and boulders. When natural processes such as flooding, sedimentation and the recruitment of large woody debris are functional, these processes create and maintain aquatic ecosystems over time. Before the Bureau of Fisheries surveys, there had been no large floods (>50-yr return interval) in the Columbia River Basin since 1894, and the longest drought on record occurred during 1928–1941 (U.S. Geological Survey 1991). Since the completion of the Bureau of Fisheries surveys in 1945, two large floods (in 1948 and 1964–65) have affected major portions of the Columbia River Basin (U.S. Geological Survey 1991).

Management emphasis clearly affected the magnitude and direction of change in pools; nevertheless, some natural streams lost pools, and some commodity streams gained pools. Natural streams could lose pools in the short-term as the result of stochastic events; however, over time, pools should reform if forming features are still functional. We hypothesize that some commodity streams may still support high-integrity habitat because of a lag time between human activities and detectable effects, or that some watersheds may be inherently more resistant than others to human activities. Lee et al. (1997) conclude that, while natural streams are significantly more likely to provide high-quality fish habitat and support strong populations, these characteristics are not necessarily excluded from commodity streams. We have an opportunity to learn from these commodity yet high-integrity streams, and we cannot assume that because some commodity streams maintain good habitat and strong populations it is prudent to actively manage roadless areas (i.e., timber harvest and road construction). The key to advancing our knowledge of these potential relationships is to determine what is different about these higher quality commodity streams and to apply these findings to watershed restoration and future human activities.

Our analysis also showed a regional pattern to change. All ecoregions except the North Cascades ecoregion showed significant decreases in pools, particularly in commodity streams. The increased pool frequencies in the North Cascades ecoregion occurred despite human activities, although increases in pools were twice as large in natural streams as in commodity streams. Prior research (Mullan et al. 1992, McIntosh et al. 1994*b,* Wissmar et al. 1994*b,* McIntosh 1995) suggests that the land use history in the North Cascades ecoregion was different from that in the rest of the Columbia River Basin. This region was removed from the major settlement routes, such as the Oregon Trail, and remains sparsely populated today. Timber harvest did not begin in earnest until the late 1960s, and land allocation in the region is unique. More than 65% of the land base of the Wenatchee and Okanogan National Forests is in roadless or wilderness areas, effectively protecting the headwaters of most watersheds (McIntosh et al. 1994*b*).

These results reinforce the idea that management emphasis and land allocation within a watershed is critical to the protection and restoration of aquatic ecosystems. Pool frequencies consistently remained the same or increased within designated wilderness and roadless areas, such as the Middle Fork Salmon River. These conclusions have important implications in the ongoing debate over how to protect and restore the rivers and watersheds of the Pacific Northwest. Current approaches being implemented by the Federal Government, such as Forest Ecosystem Management Assessment Team (FEMAT; U.S. Department of Agriculture 1993), have adopted key watersheds and the use of large riparian reserves for aquatic ecosystem management, protection, and restoration. An unresolved question is whether key watersheds and riparian reserves have characteristics similar to those of large, naturally functioning watersheds. A critical caveat may be that these watersheds are similar only if they are allowed to recover and function naturally. This means regulating activities (e.g., new roads, timber harvest, livestock grazing) in these watersheds so as not to impede or forestall recovery processes.

Our analysis of land use records showed regional patterns of Euro-American development in the Columbia River Basin. Development generally proceeded up the basin and along the major migration routes. Areas of greater isolation, such as the North Cascades ecoregion (Mullan et al. 1992) and the Northern Rockies ecoregion (Lee et al. 1997), remain relatively undeveloped today. Early development pressures focused around population centers and readily accessible areas (e.g., along waterways). As populations grew, demand for resources caused development to expand throughout the watershed. By the turn of the century, rangelands were severely overgrazed, and many stream/riparian systems had been simplified by snagging, log drives, splash dams, timber harvest, and mining. The livestock industry grew rapidly over the settlement period, but changed from a mix of cattle and sheep to today's domination by cattle. Current livestock use in the Columbia River Basin is at the highest level since settlement began.

After World War II, the availability of heavy equipment and increased road construction allowed the timber industry and the U.S. Forest Service to expand into previously inaccessible areas. Intensive timber harvest and road construction continued until the late 1980s, when concerns for endangered species and old-growth forests slowed harvest. The boom in the timber industry further simplified stream/riparian ecosystems by increasing sediment delivery and peak flows, reducing or eliminating the interaction between stream channels and floodplains, and reducing large woody debris and riparian vegetation. Fisheries professionals also played a role in the decline in stream habitats by recommending the removal of large woody debris to reduce barriers to fish migration. These practices acted cumulatively to reduce the capacity of stream ecosystems to recover from disturbance, either natural or anthropogenic, and to support self-sustaining fish communities. As Beschta et al. (1995) concluded, the legacy of past practices already limits the function and integrity of existing watersheds. Today's managers must not only manage for current needs, but must also correct the mistakes of the past.

We found only one study that potentially contradicted our findings. Carlson et al. (1990) concluded that past timber harvest practices had not altered stream habitats in northeast Oregon streams. They examined the effects of timber harvest in stream segments where harvest had occurred in the past 6–17 years. Their study streams consisted of short reaches (300 m) in small watersheds (drainage area $\leq$25 km^2). Roads were not located in the stream/riparian corridor, but were typically along ridges, and skid trails were properly drained. We conclude that the land use history for these stream segments is not representative of streams in the rest of Blue Mountain ecoregion or the Columbia River Basin. Unlike our study streams, these stream segments had not been affected continuously since the beginning of Euro-American development by grazing, splash dams, log drives, repeated entry and harvest, and roads. The lack of a response in their study streams could also be due to a lag time between activities and effects, or these streams may be in lithologies that are more resistant than others to human activities.

Our results provide unique opportunities for future research. If current attempts at watershed analysis by land management agencies continue (U.S. Department of Agriculture 1993), our collective understanding of the effects of disturbances may improve. As watershed analysis is currently posed, it may provide a systematic method for characterizing watershed conditions, along with the ecological and watershed-related processes that determine the biophysical capabilities of a watershed (U.S. Department of Agriculture 1993). Because of the lack of data and knowledge about how some disturbances affect streams, we were unable to quantify the magnitude and extent of all anthropogenic and natural disturbances likely to affect stream ecosystems. We were able to document regional trends only in readily quantifiable land uses such as timber harvest and grazing; little data were available for assessing fire or floods. Developing cause-and-effect relationships between changes in pools and disturbance, both human caused and natural, is likely to provide important information that could lead to a stronger ecological approach to land management and restoration than were past efforts.

What do these changes mean for fish and fish habitat? We would conclude that the capability of streams in commodity watersheds in the Columbia River basin to support fish and other aquatic organisms has been severely reduced. Besides the loss of pool habitats, common characteristics include high stream temperatures, fine sediment levels, and low large woody debris levels (Lee et al. 1997). These conditions act cumulatively to simplify stream habitats available to the diversity of native fishes endemic to the region. These conclusions are magnified by the status of anadromous fishes in the Columbia River basin. Most of the anadromous fish fauna in the basin is now at risk of extinction (National Research Council 1996, Thurow et al. 1997). A guarded exception is the North Cascades ecoregion, where most native fish species are listed as depressed but stable (National Research Council 1996). We believe it is no coincidence that this corresponds to the ecoregion where water temperatures are still relatively cool and where pool habitats have increased over the last 50–60 yr.

This conclusion is especially intriguing, given that anadromous fish from the North Cascades ecoregion must pass four to nine main stem dams each direction over their lifecycles. In the Snake River Basin, anadromous fishes must pass four to eight main stem dams, yet most anadromous stocks from the Snake are either endangered or severely depressed. Improved habitat conditions in the North Cascades Ecoregion may have slowed the declined of anadromous fishes compared with other Ecoregions.

A final consideration is the location of these streams on the landscape. Most of the managed streams in the Bureau of Fisheries survey occurred in the lower reaches of the watershed. The focus of their study was streams that had supported spring chinook salmon. These portions of the watershed are not currently a major part of the debate over salmon, let alone watershed restoration. The focus instead is the upper reaches where salmonids still survive. In addition, headwater reaches are typically on public lands, where restoration may be easier due to ownership and continuity. These headwater streams have experienced major improvements in Federal Land management in the last decade (U.S. Department of Agriculture 1993, U.S. Forest Service and U.S. Department of the Interior 1995). Lower reaches are not given much priority, due to the complexities of private ownership and the fact that most of these reaches are currently uninhabitable by salmonids. Management of riparian areas and fish habitat on private lands is voluntary and often deals with short-term fixes such as in-stream structures, instead of long-term riparian and watershed restoration. The emergence of watershed councils within the Basin offers some hope that habitat restoration can be strategically focused to greatest advantage for the fish. These data are important in aiding these councils in defining their future vision of stream habitats.

Lichatowich and Mobrand (1995) have argued that these lower reaches were where most chinook salmon production came from historically. They proposed that focusing habitat restoration on the upper reaches will bring minimal gains in chinook salmon populations. Instead, restoring habitats in the lower reaches and connectivity between the lower and upper reaches of watersheds is essential to the revival of chinook salmon populations (Lichatowich and Mobrand 1995). We agree with these conclusions. In the short-term (<10 yr), our best opportunity to slow the decline is on public land. Over the long-term, we cannot reconnect watersheds and restore salmon without significant contributions from private lands.

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We dedicate this research to all the individuals who worked on the Bureau of Fisheries Survey from 1934 to 1945. They toiled many long hours under difficult circumstances, only to have their data sit in boxes for 50 years, virtually unknown to the rest of the world. We are very proud to have followed in their footsteps in attempting to bring this work to life. A special commendation to the late David Frey for visiting Corvallis and reliving his memories of working on the Bureau of Fisheries survey with us. We also thank Doug Dompier and Dale McCullough of the Columbia River Inter-Tribal Fish Commission for helping us track down the original surveys in 1987, saving them from certain destruction.

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