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An approach to effectiveness monitoring of floodplain channel aquatic habitat: channel condition assessment

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Abstract

The condition of aquatic habitat and the health of species dependent on that habitat are issues of significant concern to land management agencies, other organizations, and the public at large in southeastern Alaska, as well as along much of the Pacific coastal region of North America. We develop and test a set of effectiveness monitoring procedures for measuring change in floodplain channel habitat in southeastern Alaska. Variables include measures of channel morphology, pool size, pool spatial density, and bed surface grain size distribution. These procedures provide methods of data collection and analysis that, in the context of a statistically defensible sampling protocol, allow for determination of rate and direction of change among different intensities of land use, and thereby evaluation of management strategies. Assessment of channel condition can also contribute to evaluation of both restoration needs and success of restoration activities. Information gained from these procedures, together with information, where available, on watershed and riparian condition and processes and land use history will contribute to interpretation of measured change and its linkage to specific disturbances. Relationships among channel condition indicators and salmonid densities as well as opportunities for future research to better understand ecosystem elements that support biologic productivity are addressed in a companion paper in this volume (Bryant and Edwards).

Keywords: Channel condition; Aquatic habitat; Effectiveness monitoring

1. Introduction

1.1. Background

The condition of aquatic habitat and the health of species dependent on that habitat are issues of significant concern to land management agencies, other organizations, and the public at large in southeastern Alaska, as well as along much of the Pacific coastal region of North America. Federal government responses to concerns over habitat degradation and species decline include, among other things, legislation, e.g., the Endangered Species Act, and species and water quality recovery plans. At the national forest scale, standards and guidelines, intended to protect aquatic habitat, are included in forest land and resource management plans.

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The current land management plan for the Tongass National Forest (TLMP) in southeastern Alaska calls for "the maintenance or restoration of the natural range and frequency of aquatic habitat and stream channel and bank conditions", and the question is posed, Are fish and riparian standards and guidelines effective in maintaining or improving fish habitat (USDA, 1997)? An answer to this question requires an effectiveness monitoring program based on procedures that allow the existing state and variability of channel condition to be objectively and precisely measured to quantify changes over time within and among channels. We use the term "effectiveness monitoring" to mean quantitative monitoring of the effectiveness of a suite of land management practices at achieving stated goals.

Quality of stream habitat is constantly changing in response to background and anthropogenic disturbances, and monitoring change by using variables that are sensitive to geomorphic processes will improve our understanding of habitat sensitivity to such disturbances. Geomorphic systems are dynamic, and both structure and process can be complex (Chorley et al., 1984). Stream habitat is structure created, maintained, and disturbed by spatially and temporally varying geomorphic processes, which force change in channel condition and habitat quality (Benda et al., 1998). Although channel types differ in their susceptibility to structural change (Schumm, 1977; Montgomery and Buffington, 1998), in alluvial, bar-pool streams, as addressed in this study, there is no basis for expecting channel condition to be static. Rather, channel condition is constantly changing in response to and recovery from background and anthropogenic disturbance. Anthropogenic effects can alter the frequency of geomorphic processes (Richards, 1982), resulting in alterations to the rate of background habitat change. Recognizing the dynamic nature of stream channel structure and process (Dury, 1966; Schumm, 1985), the resulting variability in channel condition (Hack and Goodlett, 1960; Montgomery and Buffington, 1998; Buffington et al., 2002a), and the implications for fish habitat (Frissell et al., 1986; Bisson et al., 1997; Reeves et al., 1998), it is clear that an understanding of the link between geomorphic processes and channel condition will facilitate interpretation of habitat quality.

Monitoring change in channel condition by using variables that are sensitive to geomorphic processes will improve our understanding of habitat sensitivity to disturbances, including land use practices. Effectiveness monitoring procedures provide methods of data collection and analysis that can, provided sufficient data, lead to determination of the rate and direction of change in channel condition across multiple types and intensities of land use. This information is valuable for evaluation of management strategies. Assessment of channel condition can also contribute to the evaluation of both restoration needs and success of restoration activities.

1.2. Context

The large-scale spatial context for issues related to aquatic habitat in southeastern Alaska is the Pacific coastal region of North America, where aquatic species, including anadromous salmonids, are an important resource in decline, and habitat alteration is one of several factors that play a major role in the decrease in these populations. Several recent syntheses address these issues (Salo and Cundy, 1987; Poff and Ward, 1990; Meehan, 1991; Bisson et al., 1997; Gregory and Bisson, 1997; Nehlson, 1997; Stouder et al., 1997; Naiman and Bilby, 1998; Halupka et al., 2000).

Large expense goes into stream habitat monitoring within this region. Johnson et al. (2001) review 112 documents containing 429 monitoring protocols that are relevant to salmonids, primarily in Washington, Oregon, British Columbia, and the northern Rocky Mountains. To set the context and clarify the scope of this paper, we briefly mention the largest of these monitoring efforts. In the early 1990s, the U.S. Environmental Protection Agency (EPA) initiated the Environmental Monitoring and Assessment Program (EMAP). Objectives of EMAP include, among others, the estimation of current status, trends, and changes in indicators of the Nation's ecological resources, including lotic habitat, and understanding associations between indicators of background and anthropogenic stresses and indicators of the condition of ecological resources (Messer et al., 1991; USEPA, 1997). To date, thousands of streams have been sampled throughout much of the country as part of EMAP (Kaufmann et al., 1999). Another nationwide monitoring effort, fully implemented in the early 1990s is the U.S. Geological Survey's (USGS) National Water-Quality Assessment Program (NAWQA). Objectives include the description and monitoring of changes in current water quality (including habitat) in a large part of the Nation's freshwater streams and aquifers and understanding background and human factors affecting water quality (Fitzpatrick et al., 1998).

At the regional scale, the USDA Forest Service (FS) in collaboration with USGS, EPA, National Oceanic and Atmospheric Administration (NOAA) Fisheries, and USDI Bureau of Land Management (BLM) is developing the Aquatic-Riparian Effectiveness Monitoring Plan (AREMP). The AREMP outlines strategies for monitoring the effectiveness of the Northwest Forest Plan's aquatic conservation strategy on federal lands in large portions of Washington, Oregon, and northern California (Reeves et al., 2004). Also at the regional scale, the FS is cooperating with BLM, National Marine Fisheries Service, and the U.S. Fish and Wildlife Service to develop an effectiveness monitoring plan to evaluate the effects of land management on watershed condition and aquatic and riparian habitat within large areas of the upper Columbia River basin in Washington, Oregon, Idaho, and Montana (the area of "PACFISH, INFISH, and the NOAA Fisheries, Columbia River Biological Opinion" ("PIBO")) (Kershner et al., 2001).

Relative to these national and regional efforts, the study reported herein is more focused in scope, both spatially and with respect to types of channels measured and number of streams and habitat variables addressed. These effectiveness monitoring procedures are intended to be feasible for application by a single agency unit, in this case the Tongass National Forest. Although we do not attempt to forecast costs of implementing these procedures, the number of proposed variables and measurement intensity are intended to be realistic given a general sense of the budgetary resources commonly available for monitoring on a single national forest. In contrast to the much larger scale efforts mentioned above, attention to sampling efficiency leads us to focus on a limited number of geomorphic indicator variables that are known to be sensitive to land use effects and can be measured efficiently. We also emphasize measurement precision in the selection of variables, recognizing the need for confidence in statements of change in habitat, considering effect size of the treatment being monitored, variation inherent in the sample, and measurement error (inclusive of sampling error) (MacDonald et al., 1991; Conquest and Ralph, 1998).

1.3. Objective

Our goal is to develop an approach and evaluate procedures for use in low-gradient, floodplain channels (Dunne and Leopold, 1978) in southeastern Alaska to measure change in stream habitat condition. We do not attempt to provide a rigorous, regionwide analysis of existing channel condition status or response to disturbance. Furthermore, we do not attempt to present a complete monitoring plan. Rather, we develop and evaluate procedures that could be applied to a variety of current or future assessments, irrespective of past, current, or future management scenarios. Our objective is to develop, test, and refine application and analysis procedures for effectiveness monitoring of floodplain channel condition in southeastern Alaska. These procedures, when applied within a framework of a statistically defensible sampling design, will provide tools to help land managers determine the effectiveness of management standards and guidelines. This objective has three components:

(1) Based largely on previous research, select variables that are the most likely to be successful indicators of change in channel condition.
(2) Develop, test, and refine field procedures for objective, precise, and efficient measurement of these variables.
(3) Demonstrate analysis procedures to test channel condition variables for evidence of response to land use.

1.4. Previous studies

In lower gradient (less than about 0.025 m/m) alluvial streams in particular, several indicators of channel condition are sensitive to processes occurring in the watershed. Previous studies provide insight into which of these response variables may be useful for assessing the effects of land use on channel condition and the relationship of these variables to relevant processes (Sullivan et al., 1987; Chamberlin et al., 1991; Bisson et al., 1997; Montgomery and Buffington, 1998). A few of the more relevant studies are presented here.

Monitoring change in streambed morphology through analysis of repeated bed elevation surveys is a widely accepted technique with a long history of application in fluvial geomorphology (e.g., Lisle, 1981).

In forest streams the influence of flow obstructions such as large woody debris (LWD) and wood-defended banks adds complexity to forest channel morphology, processes, and related effects of land use (Keller and Swanson, 1979; Buffington et al., 2002b). Surveys of bed topography have been used in these streams to demonstrate dependence of local bed morphology and sediment storage on characteristics and stability of in-channel obstructions (Heede, 1972; Lisle, 1986a; Smith et al., 1993). Olson-Rutz and Marlow (1992) present a technique for analyzing magnitude of change in cross-sectional area.

Comparison of channel hydraulic geometry, including flow width, depth, velocity, friction factor, and width-to-discharge ratios (Leopold and Maddock, 1953) to a control has been applied to assessment of channel condition response to land use. Lisle (1986b) analyzed the effects of LWD on hydraulic geometry, pool characteristics, and storage of fine sediment in eight forest streams on Prince of Wales Island, southeastern Alaska, along reaches with either pristine or logged riparian areas. Velocity was significantly less and depth and friction factor were significantly greater at a particular discharge in the logged streams, owing to greater LWD loading. Logged streams had larger percentages of the bed surface covered by fine sediment, presumably owing to greater hydraulic roughness (see also Buffington and Montgomery, 1999a) and more abundant low-energy environments, caused by greater LWD loading (see also Smith et al., 1993). There was no significant difference between reaches in pristine and logged areas with respect to number of pools, distribution of residual pool depth (Bathurst, 1981; Lisle, 1987), or width-to-discharge relations (Lisle, 1986b).

Hogan and Church (1989) analyzed hydraulic geometry to quantify hydraulic characteristics and predict availability of salmonid habitat in a logged and a pristine stream reach in the Queen Charlotte Islands, British Columbia. They found that flow in the logged channel tended to be wider, shallower, and faster than predicted from hydraulic geometry-drainage area relations and attributed this to land use impacts. These effects resulted in less than the predicted area of the channel being hydraulically usable for salmonids (Hogan and Church, 1989). Earlier work in two nearby pairs of basins indicated that logging was associated with increased channel width and riffle area and decreased pool area (Hogan, 1987).

Size and distribution of channel habitat units, e.g., pools and riffles, and LWD are commonly used as indicators of channel condition, particularly with respect to characterizing aquatic habitat (Bisson et al., 1981; Hankin and Reeves, 1988). Channel units are basic morphological components of stream reaches (Leopold et al., 1964), generally 0.1–10 channel widths in length. They are commonly divided into types based on physical and hydraulic characteristics (Bisson et al., 1981; Sullivan, 1986).

Carlson et al. (1990) compared channel features in five relatively pristine stream segments in northeastern Oregon with paired segments having one-quarter to one-half of their riparian forest removed. They found no significant difference between logged and pristine pairs with respect to number of pools per 100 m or percentage of stream area in pools. In a comparison of 70 forest stream reaches in pristine, recently clearcut, and second-growth areas in southwestern Washington, Bilby and Ward (1991) found significant differences in the frequency of LWD-related pools. For a given channel width, pristine reaches had the highest frequency while clearcut reaches had the lowest. Channels in the pristine areas had a much broader diversity of pool types. The LWD loading was also significantly different among land use intensities, with pristine and second-growth areas having the highest and lowest loadings, respectively (Bilby and Ward, 1991).

Reeves et al. (1993) examined timber harvest effects on pool frequency, wood loading, and diversity of salmonid populations in 14 coastal Oregon channels. Difference in pool frequency between basins with low and high (>25% basin area) timber harvest intensity was not consistently statistically significant. Both LWD loading and salmonid diversity were significantly less in basins with high harvest levels. Ralph et al. (1994) compared unharvested, old-growth forest streams to those in moderately and intensively logged basins in western Washington state. They found no differences in frequency of LWD pieces, although LWD was smaller and concentrated along channel margins, and pool area and depth were reduced in intensively harvested basins (Ralph et al., 1994). Montgomery et al. (1995) found that clearcut timber harvesting was associated with reduced LWD loading and thus lower pool frequency.

Streambed grain size distribution (texture) may be useful as an indicator of channel and watershed condition (Bunte and Abt, 2001). Texture responds to changes in sediment supply (Dietrich et al., 1989; Lisle et al., 1993; Buffington and Montgomery, 1999b), as does the volume of fine sediment stored in pools (Lisle and Hilton, 1999). Buffington and Montgomery (1999a) analyzed streambed surface grain size distribution in gravel-bed rivers, including sites in southeastern Alaska. They found that grain size was responsive to hydraulic roughness caused by bank irregularities, bars, and LWD. Channels with greater hydraulic roughness had finer grained bed surfaces, presumably because fluid energy was extracted by roughness elements, thereby reducing energy available for bed-load transport. Resulting textural fining provided usable salmonid spawning habitat in channels that would otherwise be too coarse grained (Buffington and Montgomery, 1999a).

Several studies have examined the effects of land use on multiple habitat quality indicators. For example, Murphy et al. (1986) examined relationships among salmonid population densities in old-growth, buffered, and clearcut channels in southeastern Alaska and a number of indicators, including percentage of fine sediment, pool and LWD volumes, standing crops of periphyton and benthos, and others. Treatment effects were inconsistent among blocks. Clearcut reaches had smaller pool and LWD volumes than found in old growth, and juvenile coho salmon abundance was directly related to LWD volume. No differences were found, however, in percentage of fines. Buffered reaches had larger volumes of LWD than found in old growth (Murphy et al., 1986).

A well-known experiment was conducted in Carnation Creek, British Columbia, from 1970 to 1987 (Hartman and Scrivener, 1990). Timber harvesting and associated roading were conducted at three levels of intensity including clearcutting to the streambank with significant in-channel felling and yarding, clearcutting to the stream margin with virtually no in-channel activity, and a variable-width (1–70 m) buffer strip treatment. Repeat channel mapping indicated that in the two more intense treatments, stability of LWD decreased, and following logging, LWD volume was reduced to about 30% of the pre-logging level, whereas no change occurred in the buffered treatments. Coincident with these changes in LWD distribution, channel width increased significantly only in the more intensive treatments, up to 8 m of bank erosion occurred, and pool depths decreased. Following logging, streambed scour and fill and bed-load transport increased. Frozen core samples of the streambed indicated that fine sediment tended to increase throughout the study period (Hartman and Scrivener, 1990).

Woodsmith and Buffington (1996) employed multivariate statistical analyses of geomorphic variables including channel morphology, channel unit size and distribution, and substrate characteristics from 23 forest stream reaches in southeastern Alaska to test discrimination of pristine from intensively harvested channel conditions. They limited measurement error by adhering to a strict protocol and including only variables that could be measured objectively and with reasonable precision. They demonstrated a minimum 90% correct classification of stream reaches into these two end-member categories of land use intensity. Their analyses identified the following objective and repeatable (when unambiguous definitions are applied) measures of physical channel condition as the most successful for distinguishing these distinct channel conditions: pool spatial density, the ratio of mean residual pool depth to mean bankfull depth, and the ratio of the median surface grain size to that theoretically predicted for bankfull flow. The authors predicted that with larger sample sizes, other variables such as channel width-to-depth ratio and relative roughness might also be useful for discriminating channel condition (Woodsmith and Buffington, 1996).

Assurance of data quality is essential to an effectiveness monitoring program (MacDonald et al., 1991; Bauer and Ralph, 1999), yet channel condition assessment can be subject to considerable measurement error (Platts et al., 1983; Ralph et al., 1994). For example, variance calculations in the commonly employed basinwide visual estimation technique infer zero error in channel unit classification and direct measurements (Dolloff et al., 1993); however, this is not the experience of many workers in the field. Large measurement error implies the risk that interpretations and decisions may be made on the basis of measurement artifacts rather than accurate data (Conquest and Ralph, 1998). Kaufmann et al. (1999) find that variability among crews is a serious concern, requiring oversight and careful training.

Large error in channel unit classification and inventory commonly results from lack of application of objective definitions of channel units, variation in measurement techniques, and disregard for changes in stream discharge (Ralph et al., 1994; Roper and Scarnecchia, 1995; Wang et al., 1996). Roper and Scarnecchia (1995) examined variability in habitat survey results by using either six or eight trained observers in three different trials. In a trial classification of nine secondary channel units by eight independent observers, classification was unanimous for only one unit. Other units were classified into as many as five different habitat types by the eight observers. A more uniformly trained group of six observers agreed on 73% of primary channel unit classifications, but on only 23% of secondary classifications (Roper and Scarnecchia, 1995). Wang et al. (1996) examined accuracy and precision of stream habitat variable estimates from three streams. For six observers, magnitude of the 95% confidence interval about the mean for estimates of percentage of area in each primary channel unit ranged from 12% to 117% of the mean. Two-thirds of the confidence interval magnitudes for various channel units were between 26% and 43% of the mean. Poole et al. (1997) concluded that commonly used habitat classification procedures were inappropriate for monitoring aquatic habitat. These procedures generally lacked the necessary repeatability and precision to detect important change, were difficult to transfer effectively among observers, and could be insensitive to anthropogenic effects (Poole et al., 1997). Peterson and Wollrab (1999) examined fish habitat inventory procedures in use by the USDA Forest Service in the intermountain Western United States and determined that procedures were subjective, biased, and inadequate for monitoring because these procedures could not reliably detect habitat change. Deficiencies included lack of consistency in measurement, inadequate QA/QC procedures, and bias in reach selection (Peterson and Wollrab, 1999).

Kaufmann et al. (1999) analyzed precision of habitat survey methods used by EPA in EMAP in several hundred streams in Oregon and the mid Atlantic region between 1993 and 1996. The signal-to-noise ratio (S:N) of variables was calculated as the ratio of within-year variation among streams, not attributable to measurement error or interannual variability, to the pooled variances of within-season repeat visits to sites. This "noise" represented within-season habitat variation and differences among crews. Kaufmann et al. (1999) described $S{:}N < 2$ as imprecise, yielding distorted estimates of indicator variables, while for $S{:}N > 10$, short-term temporal variance and measurement error caused relatively insignificant error in estimates of indicator variables. They found that measuring areal percentage in habitat types was imprecise, varying with stage and among observers (see also Platts et al., 1983). In Oregon streams, percentage of area in pools had a signal-to-noise ratio of 2.1. Percentage of specific types of pools was particularly imprecise ($0.1 < S{:}N < 2.5$). In contrast, signal-to-noise ratio for mean residual depth measurements was 9. Measures of mean bankfull width were highly precise ($S{:}N = 24$), whereas wetted width-to-depth ratio was moderately precise ($S{:}N = 6.5$) for Oregon streams. Indicators of substrate size were variable in precision (see also Wang et al., 1996). For percentage of fines, $S{:}N = 15$ and for percentage of sand, $S{:}N = 0.1$, whereas measures of central tendency of substrate size class had $S{:}N = 23$ for Oregon streams. However, many observers find that adequately characterizing substrate size distribution requires very large sample sizes (Bunte and Abt, 2001) or partitioning the bed into identifiable textural patches (Buffington, 1999). A measure of bed stability, the ratio of the median surface grain size to estimated maximum size entrained at bankfull flow had $S{:}N = 6.8$, and various measures of frequency and volume of LWD had $2.4 < S{:}N < 12$ for Oregon streams (Kaufmann et al., 1999). Ralph et al. (1994) found measures of LWD volume and position to be objective and repeatable. Improved methods of physical habitat characterization are being tested as part of the EPA's EMAP (Kaufmann and Robison, 1998).

Roper et al. (2002) investigated the variance structure of several commonly used monitoring indicators. They found that the total sample size required to detect a difference of 20% in an indicator (with Type I and Type II errors set at 0.10) was nearly 400 or greater for gradient, median surface substrate size, and two measures of percentage of fines. Furthermore, observer error exceeded 20% of total variance for percentage of fines, percentage of pools, and percentage of stable banks.

2. Methods

2.1. Approach

Assessment of channel condition status and trend can be attempted at a wide range of spatial scales depending on the information need. Bisson et al. (1997)

argue for a focus on “landscape scales large enough to encompass the freshwater life cycles of salmon and other species.” In contrast, some studies have focused on scales as small as a single reach. The work presented herein is relevant to the landscape of southeastern Alaska as represented by a spatially distributed sample of reaches. A reach is a length of stream channel with homogenous morphological, sedimentological, and hydrological features (Hogan and Church, 1989). We chose the reach as our sampling unit because it integrates very local and short-term changes occurring over multiple channel units, yet it is small enough that variables can be measured directly, avoiding error associated with visual estimation techniques (Ralph et al., 1994; Roper and Scarnecchia, 1995; Wang et al., 1996).

A likely range in channel condition parameters can be estimated from channel typing classification (Rosgen, 1994). Process-based classification approaches provide additional insights into potential response to disturbance (Schumm, 1977; Paustian et al., 1992; Whiting and Bradley, 1993; Montgomery and Buffington, 1997). Channel type classification may also provide a method to predict abundance of salmonid populations (Bryant et al., 1991). Our study is limited to channels with depositional, rather than erosional or transportational, characteristics (Schumm, 1977). Depositional reaches are likely to respond to disturbance by aggradation, degradation, or other measurable changes in channel morphology or substrate. Their typical location low in the drainage network makes them likely to integrate cumulative effects of disturbance processes occurring throughout their drainage basin. Because of this limited range in sampled channel type, differences among types are not relevant to our findings and are, therefore, not discussed. This focused approach does not provide for quantifying local responses to specific disturbances in a variety of channel types higher in the drainage network. Such an approach would require a very large increase in sampling effort and resources.

Meaningful interpretation of channel condition requires a reference standard against which to compare the state of a channel (Noon et al., 1999), irrespective of value judgments placed on a specific standard. We follow the common practice of using the central tendency and variance of pristine (apparently subjected only to background (non-anthropogenic) disturbance) stream channels as the reference condition and experimental control (Reiser and Bjornn, 1979; Sullivan et al., 1987; Conquest and Ralph, 1998).

Although many variables can be measured to characterize channel condition, it almost certainly is not necessary to measure all of them. Many are likely to be redundant, be difficult to measure with accuracy and precision, or lack a clear relationship to aquatic habitat quality. If resources for a monitoring program are limited, it is especially valuable to consider procedural efficiency. Clearly the most useful variables will be those indicative of the condition of important habitat for species of interest, most responsive to affecting land use practices, and those having variance sufficiently small that change can be detected (Bjornn and Reiser, 1991; MacDonald et al., 1991; Conquest and Ralph, 1998). Efficiency can be gained by identifying the smallest number of monitoring variables required to adequately assess channel condition.

2.2. Variable selection

Based on a review of the literature, we selected a small number of the potential channel condition variables for testing and referred to these as monitoring variables in this paper. Criteria included sensitivity to disturbance, association with important aspects of habitat quality, measurement objectivity (independence from stream discharge and other avoidable variance), precision, and efficiency. Through careful selection of monitoring variables we endeavored to minimize potential measurement error relative to actual variation in channel condition.

2.3. Site selection and characteristics

Selection of sample reaches will vary with specific monitoring objectives. We sampled 66 floodplain (Dunne and Leopold, 1978) stream reaches distributed across the landscape of southeastern Alaska. Streams were selected opportunistically, based on professional judgment, to (1) collectively represent a wide distribution in geography, physical context (geology, soils, etc.), land use intensity, and degree of recovery from disturbance; (2) be logistically feasible; and (3) where possible, be of immediate interest to land managers. Therefore, sampled streams were not randomly selected. However, reach locations within streams were randomly selected by designating the starting point of each reach at a randomly selected distance of 1–10 streambed widths from a convenient and easily recognizable landscape feature such as a stream junction or upper limit of tidal influence.

Reach length was approximately 20 channel widths. Multiple reaches were sampled in streams where site selection criteria and logistics allowed. However, sampling only one reach per stream would reduce the risk of non-independence of sample units for statistical inference. Reaches were wadable and generally single-thread channels without major (>20% of total discharge) tributaries or significant tidal, bedrock, or upstream lake influence. Evidence of bankfull elevation was required to ascertain bankfull channel geometry; therefore at least an incipient floodplain was required. Reaches generally had a bar-pool morphology largely controlled by in-channel flow obstructions, such as LWD (see also Buffington et al., 2002b). Gradients were generally <2%. Corresponding channel types included floodplain (FP) and moderate-gradient, mixed-control (MM) process groups according to the channel type classification system used by the FS Alaska Region (Paustian et al., 1992). A number of pristine reaches were included to assess background central tendency and variation in response variables.

For purposes of illustrating analysis approaches, intensity of land use affecting each reach was designated as pristine (P), moderate (M), or heavy (H). Pristine watersheds had no timber harvesting or roads, or land use was unambiguously trivial. At least one of the two conditions was required for a designation of H: (1) the product [(% watershed area clearcut) × (% riparian area cut) × (road density (m/ha))] > 0.25, or (2) % riparian area cut > 0.25. These objective limits were established post hoc to provide boundaries for classifications initially based on professional judgment. Land use intensity categories were determined for the watershed area contributing to the center point of each reach.

2.4. Data collection

We collected the following channel condition data in each reach: elevation surveys of the longitudinal and cross-sectional profiles, pool spatial density and residual depth (Lisle, 1987), substrate surface grain size distribution (Wolman, 1954), LWD inventory, and photos and sketches. We utilized the following GIS-derived data provided by the Tongass National Forest: watershed drainage area, road density, and area of timber harvesting in riparian areas and elsewhere. Watershed characterization, including riparian stand density, geology, soils, climate, and other available information will enhance interpretation of monitoring data. However, these components of watershed analysis were beyond the scope of our objectives. Field data were collected from 1997 through 2001. We also incorporated data reported by Woodsmith and Buffington (1996) and pilot study data collected by the senior author with Tongass National Forest personnel during 1993–1996.

2.4.1. Details of field procedures

These procedures were written for field personnel with a reasonable background in alluvial stream channel geomorphic survey procedures. For untrained personnel, more detailed descriptions of these techniques are readily available elsewhere (e.g., Dunne and Leopold, 1978; Platts et al., 1983; Harrelson et al., 1994). Close supervision by an experienced geomorphologist is important for keeping measurement errors within reasonable limits. Some users may conclude that additional variables and more intensive data collection are appropriate.

An initial bed width was measured near the randomly selected starting point at a place that appeared representative of the average bed width. Bed width was defined as the horizontal distance, perpendicular to the centerline of the channel, from the bottom of one bank to the bottom of the opposite bank. Bed width was measured to the nearest 0.1 m at 1-width intervals, based on the initial measurement, along the reach for a minimum of 20 widths; a mean was calculated and used for spacing cross-sections, defining minimum residual pool depth, and calculating pool density. For all purposes, left and right were based on the downstream-looking perspective.

Surveying of channel morphology followed well-established procedures (e.g., Dunne and Leopold, 1978; Harrelson et al., 1994) by using an engineering level, stadia rod, and graduated tapes or instrumentation of equal or greater accuracy and precision. When moving the level (“turning”), two turning points provided elevational control. Turns were expected to close within 2 cm or less. The longitudinal profile started at mid-channel along the first cross-section and continued along the mid-channel centerline (midway between the left and right bottom of bank). Centerline surveys were used, because, unlike thalweg surveys, they provided a close approximation of water surface slope at channel forming (bankfull) flow. Points were surveyed at important changes in bed elevation. Average distance between shots was generally 1–3 m. Cross-sectional and longitudinal survey points represented elevation of the streambed, not LWD or other material that was not representative of the surrounding bed elevation. The longitudinal profile intersection with each cross-section was surveyed (intersection distance on both tapes was noted). If the slope of the streambed changed dramatically just upstream of the upstream-most cross-section, the longitudinal bed survey was continued upstream for another five mean bed widths.

Cross-sections were located and survey monuments established perpendicular to the centerline of the channel every 5 channel widths beginning at the randomly selected starting point, resulting in 5 cross-sections per 20-width reach. Although more cross-sections would provide greater precision for channel dimension measurements, this frequency accommodated detailed instrument surveys producing accurate estimates of bankfull depth, while maintaining reasonable survey cost. Ideally, cross-sections were established across straight, uniform-flow, riffle portions of the channel; however, in forest channels the classic riffle-pool sequence is commonly disrupted by LWD or other flow obstructions, and cross-sections may need to be established at less than ideal locations. For this study, locations were moved slightly upstream or downstream where necessary to avoid deep pools or LWD jams that prohibited surveying. A minimum of approximately 20 points were surveyed along the cross-sections at important changes in bed elevation, including top and bottom (ground elevation) of cross-section monuments ("pins"), bankfull elevation (where detectable), top and bottom of streambanks, edges of stream water, thalweg, and intersection with the longitudinal profile tape (intersection distance on both tapes was noted). Determination of bankfull elevation was subject to considerable observer variation. Therefore, this elevation was determined once, generally by the principal investigator, for each cross-section and was considered fixed, thereafter.

The method of Wolman (1954) was employed to measure substrate surface grain size distribution. Where possible, these pebble counts were centered along each cross-section. If site characteristics could compromise data quality (e.g., deep water), the pebble count was shifted slightly upstream or downstream from the cross-section to a recorded location along the long profile, and future pebble counts were done at this location. A rectangular (rather than a square) grid was commonly necessary to avoid pools, log jams, etc. A minimum of 100 particles were measured by using a "gravelometer", a simple metal plate with square openings of known dimensions. At each cross-section, a total of five cross-channel traverses were made that included the entire bed width, parallel to the cross-section. Bedrock and unmeasurable, embedded particles were noted, but not counted as one of the required 100. Particles measuring <4 mm or >256 mm were recorded as 1 or 999, respectively. Combining pebble counts from each cross-section yielded a total sample of 500 particles per reach, a sample size regarded as adequate for determination of D_{50} in many gravel-bed streams (Bunte and Abt, 2001). Precision could be improved by measuring more particles, either at each cross-sectional location or at additional locations along the reach (Bunte and Abt, 2001; Kaufmann, P.R., personal communication).

The pool inventory was conducted between the upstream- and downstream-most cross-sections. Pools are defined as topographic depressions in the streambed having a residual depth (Bathurst, 1981; Lisle, 1987) equal to or greater than the value determined by the following equation: minimum residual depth = $(0.02 \times \text{mean bed width (m)}) + 0.05\,\text{m}$ and having length or width at least 10% of the mean bed width. A hand level was used in very low-gradient channels to accurately identify pool tails (hydraulic controls) for residual depth determination. Residual depths were measured and recorded. Pool-like features close to the minimum depth were measured and recorded, but not included in the analyses. Pools were classified into one of three pool types—plunge, underscour, or other (Woodsmith and Buffington, 1996). Adjoining pools were considered distinct if there was a readily detectable morphological separation on the bed between them. In practice this separation was 10 cm or more of rise in the bed. This relatively "fine-grained" approach to pool delineation incorporated morphologic complexity into the variable "pool density." All pools sharing a common tail were noted accordingly. This recording procedure facilitated data translation to a more "coarse-grained" standard if desired. All pools having a tail within the reach were included in the inventory.

For assessment of channel condition, we used LWD inventories only to characterize the reaches, rather than as a response variable, because of statistical redundancy with pool spatial density. All pieces of LWD greater than or equal to 10 cm in diameter and 1 m in length, lying between the upstream and downstream cross-sections were inventoried. At least this much of the piece (10 cm × 1 m) must have been within the bankfull channel (width and elevation) in order for the piece to be counted. The LWD type was noted as log, rootwad, log with rootwad, or other. Occasionally living trees or root masses protruded from the bank enough to scour a pool, just like an independent piece of LWD; these were counted as LWD. Under no circumstances was any piece of LWD counted twice. Number of pieces in jams or clusters of LWD were estimated if counting individual pieces could not be done precisely.

For other applications, such as examining linkages between channel structure and aquatic productivity and biodiversity, a more detailed LWD inventory may be desirable. The FS Alaska Region has developed a detailed LWD inventory procedure that is likely to meet these needs.

The entire reach was sketched, including cross-sectional and survey benchmark locations. Photos were taken and labeled from each cross-section, looking both directions across the channel and upstream and downstream. If possible, GPS readings were taken at the cross-sectional pins at both the upstream and downstream ends of the reach.

Repeat surveys of already established reaches were done between the established upstream and downstream cross-sections. It was expected that length of the reach along the channel centerline would change somewhat, owing to channel changes resulting from avulsion, bank erosion, or shifting LWD. If the stream avulsed to a new channel, the new channel was treated as a new reach in subsequent surveys.

At a few trial locations, we estimated riparian stand density by using procedures modified from Curtis (1970). Measurements were made on sample plots at the 1st, 3rd, and 5th stream cross-sections on both banks. Plots were 10 m in diameter, centered along the cross-sectional line, with the near-stream boundary of the plot at the cross-sectional pin. Plots at the 1st and 5th cross-section were moved 5 m toward the 3rd cross-section to sample only those trees within the reach. Trees that are touching the plot circumference line at their diameter at breast height (d.b.h.) were considered within the circle. Trees were categorized as live, dead standing, or dead fallen (d.b.h. was measured as if the tree were upright). The following data were recorded:

(1) d.b.h. if >4 cm (smaller trees were not recorded);
(2) mean d.b.h. for the plot;
(3) quadratic mean diameter $(Q_\mathrm{m}) = (\sum d^2/n)^{1/2}$, where d = tree d.b.h. and n = sample size;
(4) basal area (BA) = $(d/2)^2\,\pi$.

Total basal area (BA) per hectare could be used to compare stand density among reaches and over time.

Although beyond the scope of this study, watershed characterization is strongly encouraged to facilitate interpretation of monitoring data. Relevant variables include drainage area, riparian stand density, road density, area of timber harvesting, geology, soils, climate, disturbance events (e.g., landslides), and others as appropriate.

2.5. Analysis

Distribution and time series of channel condition variables were examined to compare values among land use intensity categories and assess change over time. Pristine reaches provided background control for reaches affected by land use. Reaches and levels of disturbance intensity were compared by using box-plot displays, one-way analysis of variance (ANOVA), and regression with SYSTAT 9 [1] software (SPSS, 1999). Means of channel condition variables were calculated as the arithmetic mean of the sample, and variances were calculated as $s^2 = \sum y^2/n - 1$, where y is the difference between the value of each sample element and the sample mean (Sokal and Rohlf, 1981). Procedures for these ANOVAs and simple linear regression analyses are well established and readily available in commonly used statistical references (e.g., Sokal and Rohlf, 1981; Neter et al., 1983; Zar, 1984). Common assumptions for ANOVA and regression analysis were checked by using procedures recommended

[1] The use of trade names in this paper is for the information and convenience of the reader. Such use does not constitute an official endorsement by the U.S. Department of Agriculture of any product to the exclusion of others that may be suitable.

in Wilkinson et al. (1996). Outliers were identified by examining data plots and evaluating Studentized residuals, Leverage, and Cook's distance criteria. Distributions were checked for normality by examining residuals plots and normal probability plots. Equal variance was checked by examining residual plots and applying Levene's test. Transformations were done as necessary to meet these assumptions.

In some cases more than one reach in our data set came from a single stream. This increased the likelihood that samples may not be independent. We checked for non-independence by examining autocorrelation plots of residuals, Durbin–Watson D statistics, and first-order autocorrelation values. None of these checks indicated serious non-independence of residuals employing criteria recommended by Wilkinson et al. (1996). We further checked for non-independence of residuals in the two streams with the greatest number of reaches (Maybeso Creek [9 reaches] and Trap Creek [6 reaches]) by testing for autocorrelation among reaches in regressions of the five selected monitoring variables against slope. We found no significant autocorrelations at any lag.

Differences among reaches in channel condition variables reflect differences in watershed condition, geomorphic processes, disturbance history, and climate. A statistically significant regression model of a response variable on time indicates a quantifiable temporal trend, and a significant difference in rate of change among levels of land use intensity could indicate a land use effect.

Statistical power is an important consideration in any effectiveness monitoring program. In general, power is the probability of correctly rejecting the null hypothesis in statistical inference (Zar, 1984). Power is determined by sample size, the significance criterion of the test (α), and effect size, which is the magnitude of the effect under the alternate hypothesis (Borenstein et al., 1997). In trend analysis, power is the probability of detecting a trend if the trend is actually occurring, and effect size is the magnitude of the trend to be detected (Gibbs et al., 1998).

We estimated power of ANOVAs by using SamplePower, 1.0 software (Borenstein et al., 1997). To estimate the duration and intensity of monitoring required to detect a trend in one of the monitoring variables, we employed the program, MONITOR (Gibbs, 1995). This program modeled count surveys by using Monte Carlo simulations, then generated occurrence by using regression analyses (Gibbs and Melvin, 1997). We selected α levels that balanced the risks associated with erroneously either rejecting or accepting the null hypothesis (Zar, 1984; Borenstein et al., 1997). When power was set at 90%, we set α at 0.10. Similarly, when power was set at 80%, we set α at 0.20.

Precision of commonly used channel condition variables has been analyzed in detail by others. Kaufmann et al. (1999) found that 20–50 within-season pairs of repeat samples at 8–20 sites over a period of several years were required to quantify within-season precision of habitat indicators. Such intensity of data collection was beyond the scope of this study; therefore we relied on estimates of precision presented in Kaufmann et al. (1999) and other sources, and compared these to values we obtained from nine pairs of repeat measurement visits by independent crews, closely spaced in time. For these checks on repeatability, we calculated the difference between teams as the absolute value of the difference in Y between the Team A value and Team B value relative to the Team A value $|Y_A - Y_B|/Y_A$, where Y was the reach-averaged value of each variable. Distributions and means of these differences were then calculated over the nine reaches.

3. Results

3.1. Reach characteristics

Approximately 60% of sampled reaches were located in the southern portion and 40% in the central and northern portions of southeastern Alaska (Fig. 1). Thirty-four of the 66 reaches were in the pristine, 12 in the moderate, and 20 in the heavy land use intensity categories (Table 1). Reach-mean gradients were all less than 0.023 m/m; bed surface substrates were gravel size, and channel widths ranged from 4 to 29 m (Fig. 2). These are common characteristics of low-gradient, gravel-bed, alluvial streams.

3.2. Variable selection and distribution

Based on the literature discussed previously, we selected monitoring variables that were sensitive to land use, represented biologically meaningful components of aquatic habitat, and were measurable with reason-

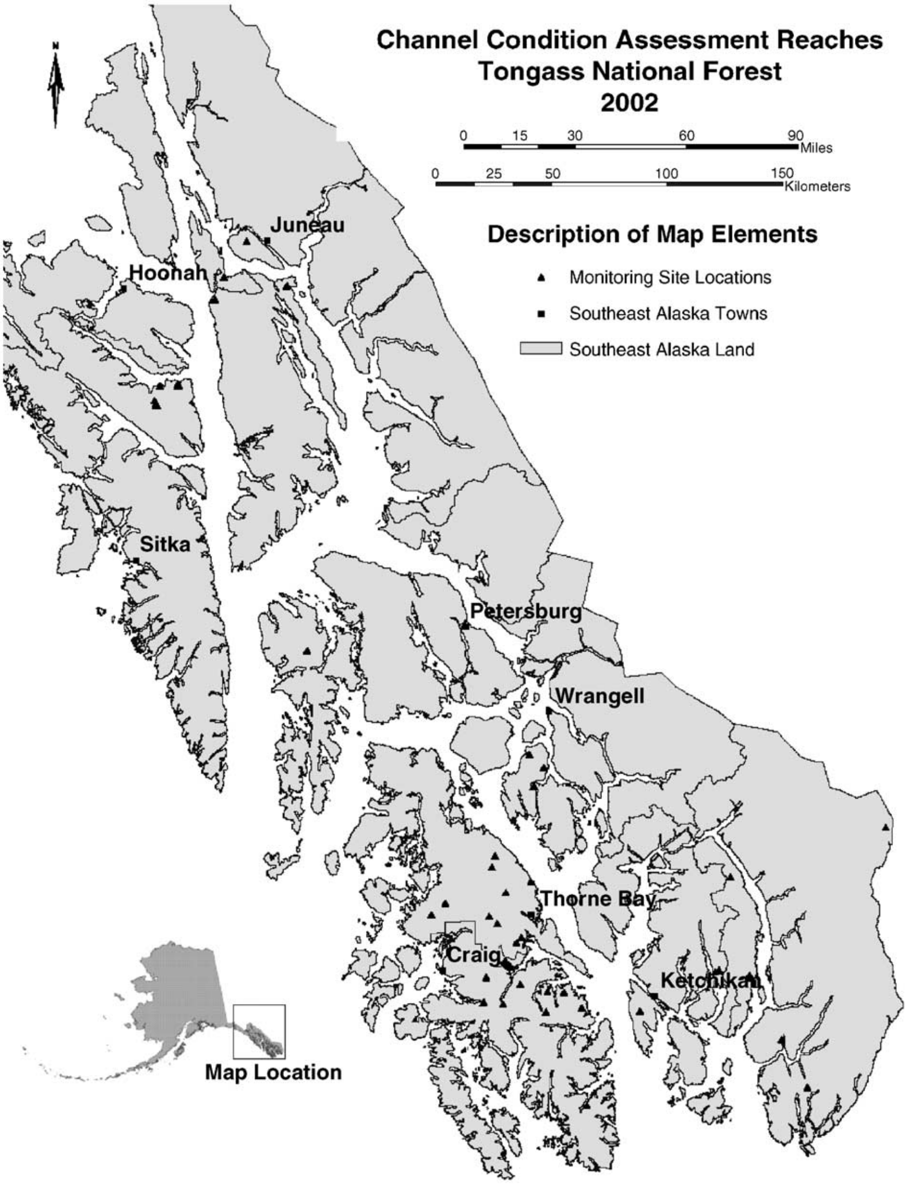

Fig. 1. Sampling site location map.

able objectivity and precision. Pool spatial density and size and substrate grain size distribution are widely recognized as important habitat components for many aquatic species. These three variables are sensitive to land use in southeastern Alaska and can be measured independently of stream discharge (Woodsmith and Buffington, 1996). We measured pool density as pools × W_{bed} (m)/L (m), the number of pools per area of channel equal to one bed width (W_{bed}) squared; L, the reach length. Pool size was measured as d_{r} (m)/d_{bf} (m); d_{r}, the reach-mean residual pool depth (Lisle, 1987) and d_{bf}, the reach-mean bankfull depth.

We represented grain size distribution of the channel bed as D_{50} (m)/$D_{50\mathrm{p}}$ (m). D_{50} is the median grain

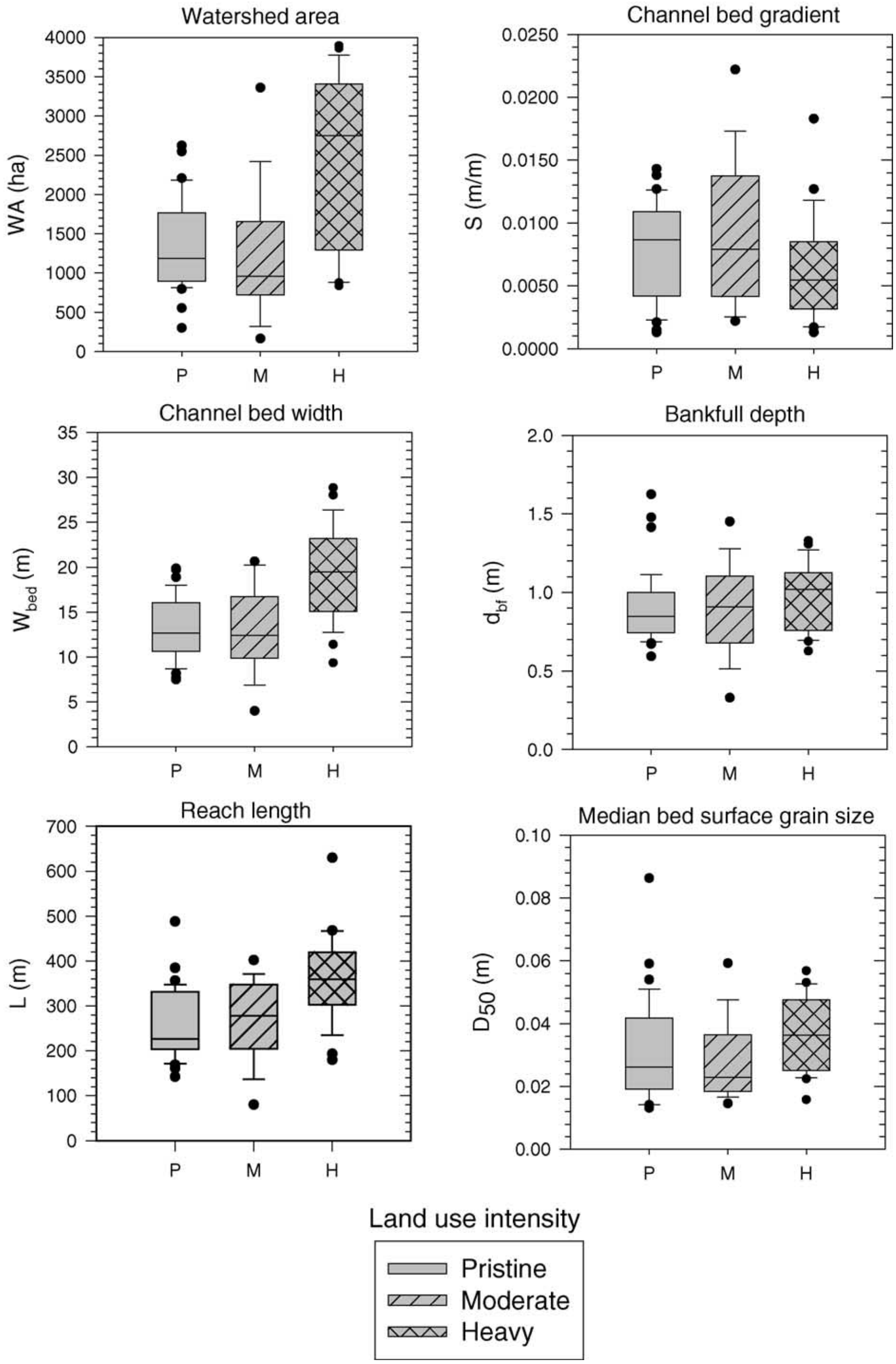

Fig. 2. Distribution of reach characteristics among land use intensity categories, as defined in the text. Box ends mark the 25th and 75th data percentiles. A solid horizontal line marks the median. Short horizontal lines mark the 10th and 90th data percentiles. Outlying data points are shown as solid circles.

Table 1
Sampled reaches

Reach	Year	Use	Reach	Year	Use
DOGS1	2000	P	BAMB	1990	M
DOGS2	2000	P	DUCK1	2000	M
DOTY1	1999	P	DUCK2	2000	M
DOTY2	1999	P	DUCK3	2000	M
EFTC	1989	P	FISH	1990, 1997, 1999	M
FOWL1	1990, 1998	P	MURI	1990, 1998	M
FOWL2	1990, 1998	P	NTHO	1996–1998, 2000	M
HOOK	1989	P	PAIN1	1996–2000	M
KADA1	1993, 1998	P	PAIN3	1996–2000	M
KADA2	1993, 1998	P	POLK	1997, 1999	M
KADA3	1993, 1998	P	SHAH	1996–1998, 2000	M
KADA4	1993, 1998	P	STAN	1996, 1997	M
KING	1996, 1999, 2000	P	12-MI	1990, 1997	H
MONI	1996, 1999	P	CABL	1990, 1997, 2000	H
OLDT	1997	P	FUBA1	1990	H
PAUL	1997	P	FUBA2	1990	H
PERK	1997, 1999	P	KDKE	1998	H
PIGG	1996, 1997	P	LUCK1	1996–1998, 2000	H
PILE1	2000	P	LUCK2	1996–1998, 2000	H
PILE2	2000	P	MAYB1	1989, 1997	H
PRIN1	1997–2000	P	MAYB2	1989, 1997	H
PRIN2	1997–2000	P	MAYB3	1989, 1997	H
PRIN3	1997–2000	P	MAYB4	1989, 1997	H
RIOR	1995	P	MAYB5	1998, 2000	H
SALT	1997, 1999	P	MAYB6	1998, 2000	H
STEP	1997	P	MAYB7	2000	H
TRAP1	1989, 1997	P	MAYB8	2000	H
TRAP2	1989, 1997	P	MAYB9	2000	H
TRAP3	1989, 1997	P	PAIN2	1996-2000	H
TRAP4	1989, 1997	P	RIOB	1995	H
TRAP5	1989, 1997	P	SAL	1996–1998, 2000	H
TRAP6	1989, 1997	P	SNIP	2000	H
WEAS1	1989	P			
WEAS1	1989	P			

Numbers in the reach name refer to multiple reaches within a stream. Years of channel condition data collection are indicated. Land use intensity is shown as pristine (P), moderate (M), or heavy (H) (see text for criteria).

size on the channel bed surface, and D_{50p} is the theoretically predicted median bed surface grain size at bankfull discharge for the specified channel geometry. Calculation of D_{50p} was based on the Shields (1936) force-balance equation relating the critical (incipient grain motion) fluid shear stress for the bed surface D_{50} to the constant dimensionless critical Shields shear stress.

Assuming a value of 0.05 for this constant, $\tau_{c50} = 0.05 \times g\ (\rho_s - \rho)\ (D_{50})$ (Vanoni, 1975), where τ_{c50} is the critical shear stress for the reach-averaged D_{50}, g the gravitational constant, ρ_s the volumetric sediment density (2.65 kg m^{-3}), and ρ the volumetric water density, Assuming that $\tau_{c50} = \tau_{bf}$, the shear stress at bankfull discharge, is approximated as $\rho g d_{bf} \cdot S$, $D_{50p} = \rho g d_{bf}\ S/0.05 \times g\ (\rho_s - \rho) = 1.0\ \mathrm{kg\ m^{-3}}\ d_{bf}$ (m) $S/0.05\ (1.65\ \mathrm{kg\ m^{-3}}) = 12.12\ d_{bf}$ (m) S.

Additional theory and rationale for this approximation are provided in Buffington and Montgomery (1999a). The ratio D_{50}/D_{50p} is similar to the "Relative Bed Stability" ratio of Dingman (1984) and mathematically equivalent to τ_{c50}/τ_{bf} employed by Woodsmith and Buffington (1996).

Width:depth ratio (W_{bed}/d_{bf}) is likewise responsive to land use (Lisle, 1982; Lyons and Beschta, 1983) and can be measured independently of discharge. We

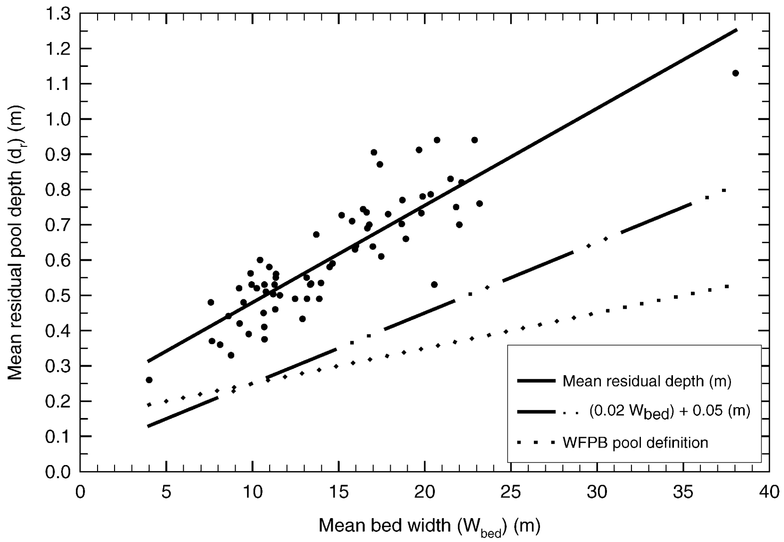

Fig. 3. Increase in mean residual pool depth with channel bed width, showing our proposed revision to the WFPB (1993) minimum residual depth definition of a pool.

used width of the channel bed (W_{bed}) for calculating this indicator and for calculating bankfull depth and other variables, thereby avoiding larger measurement error associated with identification of bankfull width. Bankfull depth was calculated for each cross-sectional survey as cross-sectional area divided by $W_{bed.}$

Relative submergence (d_{bf}/D_{50}) is related to development of bar-pool topography (Buffington et al., 2002b), and its sensitivity to land use can be inferred from the responsiveness of W_{bed}/d_{bf} and D_{50}. We used these five monitoring variables in our analyses of channel condition. The LWD frequency was excluded, owing to very close correlation, and therefore statistical redundancy, with pool density (Woodsmith and Buffington, 1996).

Lacking a widely accepted, precise definition of a pool, we initially collected data by using a channel-width-scaled definition modified from one developed by the Washington State Forest Practices Board (WFPB, 1993). According to this modified definition, to be considered a pool the minimum residual depth ($d_{r-\min}$) of a pool-like feature had to equal at least 0.01 W_{bed} (m) + 0.15 (m). This definition was similar to that used by (Woodsmith and Buffington, 1996) and has been used by the FS in southeastern Alaska. This definition was not based on rigorous analyses; rather it quantitatively described a discrimination based on professional judgment. Comparison of that minimum depth definition to the actual increase in mean residual pool depth with channel bed width indicated that the scaling factor needed to be adjusted to remove bias associated with channel size (Fig. 3). Based on these data, we found the following definition of a pool to be more appropriate for southeastern Alaska: $d_{r-\min} = 0.02\, W_{bed}$ (m) + 0.05 (m). This simplistic definition was fit "by eye" to approximate the slope of the empirical relationship and adjusted to include all morphological features considered large enough by professional judgment to be considered a pool. Like the definition it replaced, this provided a tool to promote consistency among observers at a level of precision appropriate for the available data. We avoided overfitting this relationship to allow for adjustments as the data set grows. All pool data reported herein used this definition. Adjustment of older data sets from the original definition to the proposed one was not problematic. For channels wider that 10 m, the smallest pool-like features were simply removed from the analysis. Our procedure included measurement of pool-like features smaller than the stated minimum depth; therefore we also made this adjustment without loss of data in channels 10 m or less in width.

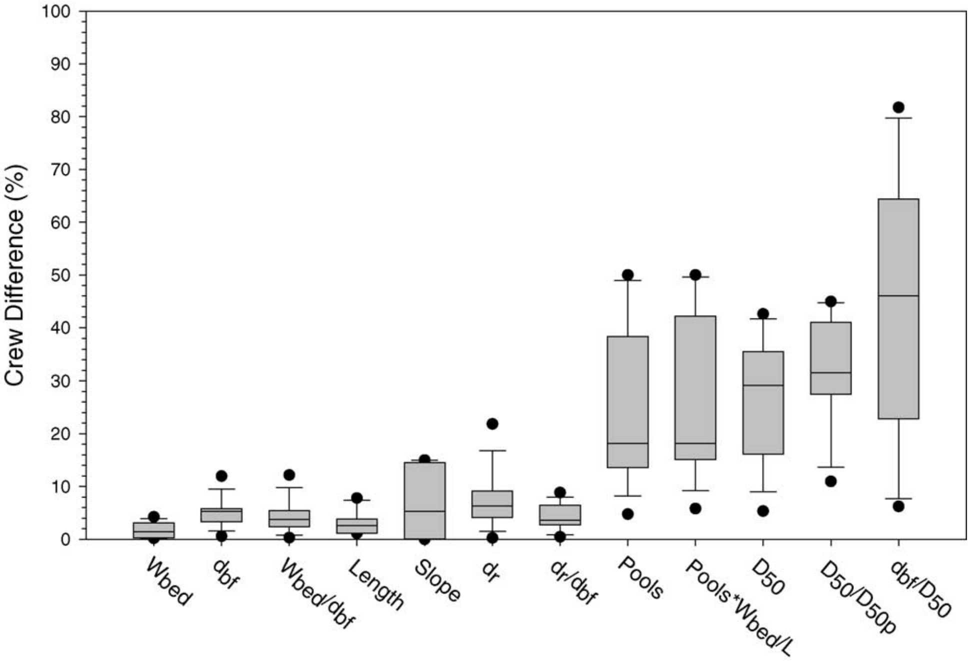

Fig. 4. Differences (%) between two crews in channel condition variable values. Box plot conventions are stated in Fig. 2. See text for explanations of variables and calculations.

3.3. Precision of variable measurement

Results of our checks on repeatability of measurements at nine reaches were in general agreement with the findings of other investigators (Ralph et al., 1994; Wang et al., 1996; Kaufmann et al., 1999), in that measurement error was lowest for surveyed measurements of channel morphology and direct measurements of pool depth. In our tests, median difference between independent teams in measured W_{bed}/d_{bf} was 4% (Fig. 4). Median differences in d_r and d_r/d_{bf} were 6% and 4%, respectively. Pool density was measured with less precision. Median differences in raw pool counts and in pool density were both 18%. Measurement of substrate grain size was least precise (see also Wang et al., 1996; Kaufmann et al., 1999; Roper et al., 2002). We found the median difference between teams in D_{50}, D_{50}/D_{50p}, and relative submergence (d_{bf}/D_{50}) to be 29%, 32%, and 46%, respectively (Fig. 4). Despite this low precision, we included measures of substrate grain size distribution in our analyses, because they were important indicators of habitat quality, and precision could have been improved by increasing the number of particles sampled (Bunte and Abt, 2001). Recent testing of EMAP methods indicated that precision of grain size estimates was substantially improved by roughly doubling the number of sampling transects in a reach, thereby increasing the number of sampled particles from 55 to 105 per reach. This focused the EMAP sampling effort on along-channel, rather than across-channel variability (Kaufmann, P.R. personal communication). However, Bunte and Abt (2001, p. 185) suggested a minimum sample size of 400 particles to limit error in D_{50} estimation to about 10%, and minimum samples of more than 1000 particles to estimate D_5 or D_{95} with the same precision. Even larger samples were suggested for fine-skewed grain size distributions, poorly sorted deposits, or sampling of reaches with multiple sedimentary units, such as pools and riffles (Bunte and Abt, 2001, p. 327).

3.4. Contrasts among categories of land use intensity

We illustrated analyses of channel condition data by using ANOVA to contrast condition of the three land

Table 2
Results of ANOVA and statistical power analyses for monitoring variables contrasted among three levels of land use intensity (P, M, and H, as defined in the text)

Variable	Probability value and (power) for Tukey multiple contrasts with $\alpha = 0.1$			Sample size for power = 0.9 and $\alpha = 0.1$ (0.8 & 0.2)		
	P vs. M	P vs. H	M vs. H	P vs. M	P vs. H	M vs. H
$\log(W_{bed}/d_{bf})$	0.979 (0.10)	0.116 (0.78)	0.312 (0.58)	>1000 (>1000)	36 (19)	37 (20)
$\log(\text{pools} \times W_{bed}/L)$	0.945 (0.10)	0.076 (0.68)	0.120 (0.53)	>1000 (>1000)	47 (25)	43 (23)
$\log(d_r/d_{bf})$	0.722 (0.21)	0.008 (0.95)	0.214 (0.55)	240 (127)	21 (11)	40(22)
$\log(D_{50}/D_{50p})$	0.962 (0.10)	0.205 (0.43)	0.246 (0.28)	769 (405)	91 (48)	103(54)
$\log(d_{bf}/D_{50})$	0.999 (0.11)	0.439 (0.54)	0.587 (0.39)	>1000 (>1000)	69 (37)	68(36)

Left-hand side: probability associated with Tukey multiple contrasts ($\alpha = 0.10$); power is given in parentheses. Right-hand side: sample sizes required from each level of land use intensity to achieve power of 0.90 ($\alpha = 0.10$) and, in parentheses, power of 0.80 ($\alpha = 0.20$).

use intensity categories, and regression to analyze and contrast trends in individual reaches. Variables were log transformed to achieve normality and equal variance; independence of residuals was verified. For these analyses, rather than follow an arbitrary convention, we selected an α level (0.10) appropriate for the high variability of the systems being studied. Watershed area had significant influence on, and was treated as a covariate of, $\log(W_{bed}/d_{bf})$ and $\log(D_{50}/D_{50p})$. There was much overlap in the distribution of monitoring variables among degrees of land use intensity, reflecting the large variability in both pristine and land-use-influenced channels (Fig. 5). Using one-way ANOVA we found no statistically significant ($\alpha = 0.10$) differences in log-transformed monitoring variables between the pristine (P) and moderate (M) or between the moderate and heavy (H) land use categories. Significant differences between the P and H categories existed for $\log(\text{pools} \times W_{bed}/L)$ and $\log(d_r/d_{bf})$ (Table 2).

Large variance in monitoring variables (Fig. 5) and limited sample size caused most of the contrasts to have inadequate statistical power to detect a difference if one existed. Estimated power of 0.9 or greater was achieved only for the P versus H contrast of $\log(d_r/d_{bf})$ (Table 2). Power of ANOVA contrasts tended to be much greater for those involving the H category (Table 2), implying that these were the reaches most affected (largest effect size) by land use. Sample sizes (number of reaches per category) were close to those required to achieve reasonable power for contrasts of width:depth ratio and pool density and size involving category H. However, much larger sample sizes would be required for contrasts of categories P versus M (Table 2, Fig. 5). Results of these contrasts and power analyses should be regarded as estimates, because streams were not randomly selected, thus introducing possible sampling bias.

3.5. Magnitude and direction of change in monitoring variables

Large spatial variation in channel condition was characteristic in both pristine reaches and those affected by land use (Fig. 5). At the beginning of this study we identified one pair of channels, Painted Creek (PAIN) (moderate to heavy land use intensity) and Princess Creek (PRIN) (pristine) for relatively intensive sampling. Time series of monitoring variables indicated considerable variability in condition of these reaches, both temporal and spatial (Fig. 6). Consistent differences in magnitude and variability among land use intensity levels were not obvious in general, although the short period of record limited definitive conclusions and the value of rigorous analyses. Clearly, short-term change in channel condition could be misleading. Temporal trends in Painted and Princess Creeks demonstrated that both magnitude and direction of change in monitoring variables could change from year to year (Fig. 6). Considering all 66 reaches in the data set and all remeasurements, magnitude of change in channel condition variables from initial reach values, averaged over each remeasurement's period of record, was large in some cases, but median values for percentage of change per year were near zero (Fig. 7).

Large variation in monitoring variables within and among reaches (Fig. 6) and generally small median effect size (Fig. 7) suggested that several years of data collection would be required to statistically verify a re-

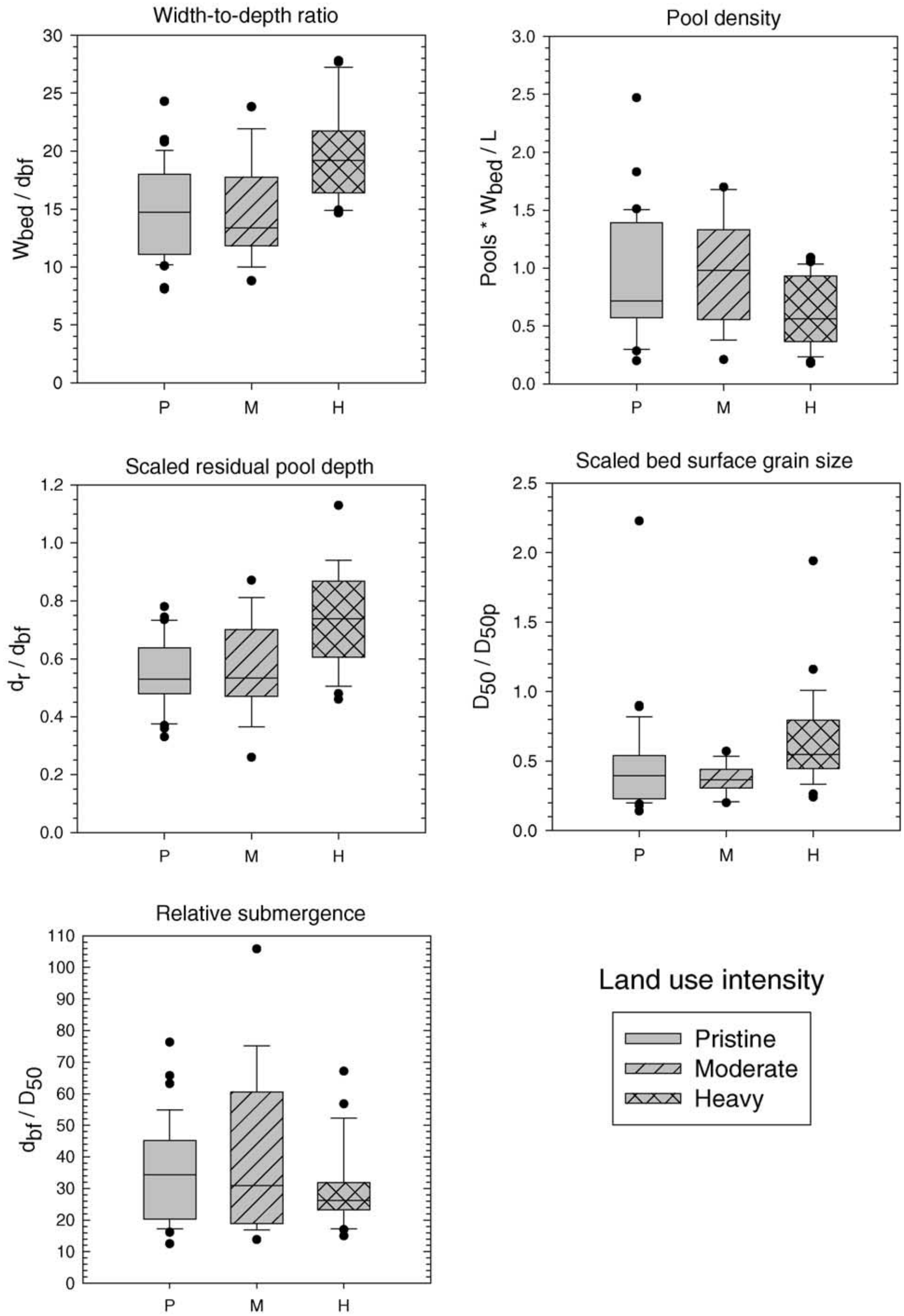

Fig. 5. Distribution of monitoring variables among categories of land use intensity. Box plot conventions are stated in Fig. 2. Variables are defined in the text.

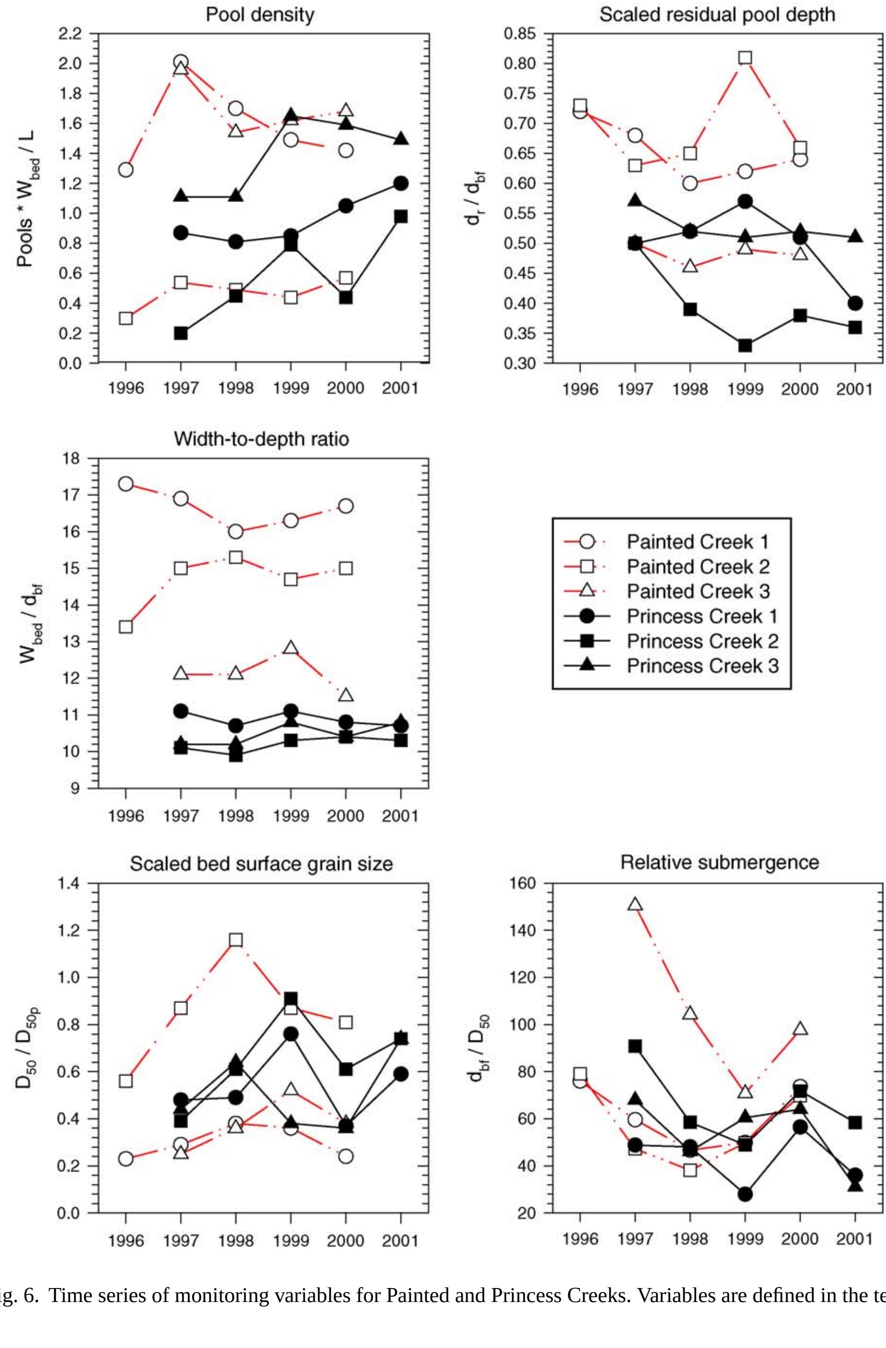

Fig. 6. Time series of monitoring variables for Painted and Princess Creeks. Variables are defined in the text.

sponse to a particular type or intensity of disturbance occurring at the watershed scale. For example, large variation, regardless of land use intensity was apparent in temporal trends in pool density in the Painted Creek and Princess Creek reaches (Fig. 8). This large variability increased the length of record necessary to arrive at statistically defensible conclusions regarding trends and differences among levels of land use intensity. Despite low statistical power resulting from the short period of record, for purposes of illustration, we

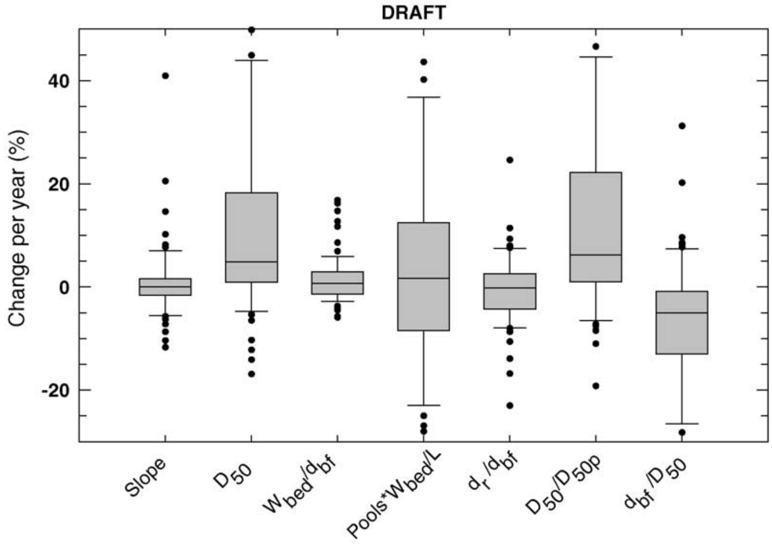

Fig. 7. Distribution of percentage of change from the initial reach-averaged value in channel condition variables considering all remeasurements in all reaches. Values are averaged over the period of record for each remeasurement. Box plot conventions are stated in Fig. 2. Variables are defined in the text.

demonstrated application of well-documented procedures for statistical testing for differences in regression models (e.g., Neter et al., 1983) by using these response variable trends. Magnitude of 95% confidence interval estimates for the PRIN1 and PAIN2 regression equations suggested that PAIN2 and PRIN2 trends were not statistically different from one another, while those for PRIN1 and PAIN2 were (Fig. 8). If statistical testing indicated that the slopes of these regressions were significant, then a trend in the monitoring variable would be verified. A statistically significant difference in rate of change (magnitude of trend) among levels of land use intensity would suggest a land use effect on the monitoring variable, although a cause-and-effect relationship would not necessarily be established.

Also for illustration purposes, we employed the program, MONITOR (Gibbs, 1995), to estimate the duration and intensity of monitoring required to detect a trend in pool density by using the data from Painted and Princess Creeks and two levels of power and associated effect size (90% with 2% annual decrease and 80% with 3% annual decrease). We presented alternate levels of power to illustrate that decisions could be made with varying degrees of certainty and a known likelihood of error. Because of the limited number of reaches with annual data, we analyzed the reaches as individual sites, rather than as part of a regional network.

The duration and measurement frequency of this subset of our data were insufficient to detect trends

Table 3
Number of years required to detect a trend in pool spatial density in Princess and Painted Creeks, given the stated power, α, and effect size (trend)

Reach	90% power, $\alpha = 0.1$, 2%/year decrease		80% power, $\alpha = 0.2$, 3%/year decrease	
	1 survey/year	3 surveys/year	1 survey/year	3 surveys/year
Princess 1	24	15	15	10
Princess 2	40	37	38	23
Princess 3	25	17	16	10
Painted 1	24	16	15	10
Painted 2	30	19	18	12
Painted 3	17	12	11	7

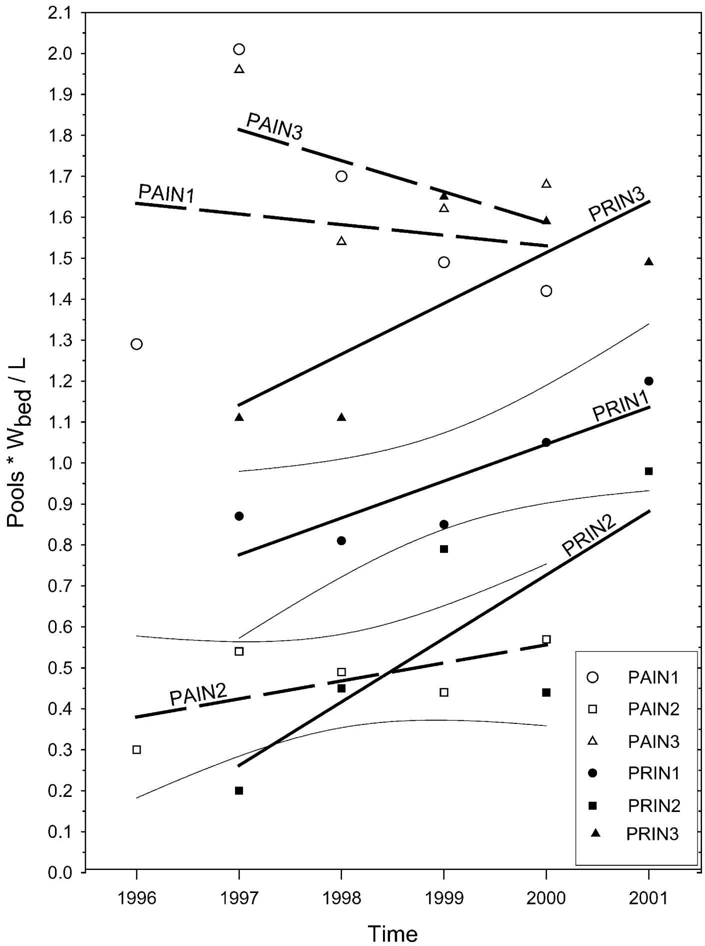

Fig. 8. Pool density trends in Painted and Princess Creek reaches. Least squares linear regression lines and 95% confidence interval estimates for PRIN1 and PAIN2 are shown for illustrative purposes only. The period of record is too short to justify statistical inference (Table 3).

in pool density. Achieving a 90% probability of detecting a 2% annual reduction would require from 17 to 40 years of annual monitoring surveys (Table 3). However, this sampling duration could be reduced by adding surveys within years, thereby reducing variance associated with measurement error (Larsen et al., 2001). With three surveys of each reach per year, the estimates of necessary monitoring duration would decline to as few as 12 years, and as few as 7 years if statistical power of 80% in detecting a 3% decrease were acceptable to decision makers (Table 3). Results of these power analyses should be regarded as estimates, because streams were not randomly selected, thus introducing possible sampling bias.

4. Discussion

The objective of this study is to develop, test, and refine application and analysis procedures for effectiveness monitoring of floodplain stream channel condition in southeastern Alaska. Objectives include neither development of a complete monitoring plan nor monitoring per se. Details of a complete, statistically defensible effectiveness monitoring plan will differ with likely land use scenarios, specific information needs, and the level of funding anticipated for the life of the plan. Close collaboration among land managers, resource specialists, researchers, and statisticians will be critically important for plan development. A first step will be to define the specific land use practices to be addressed. For example, the management information need may require quantitative comparison of only the most recent management guidelines to pristine conditions, or comparisons involving a mix of current and obsolete practices may be needed. To identify candidate sites for restoration, a focus on landscapes affected by obsolete management practices compared to pristine conditions or to the latest management techniques may be appropriate. Availability of sites affected by the scenarios of interest and commitment to long-term funding will influence whether trend detection and comparison are attempted at single, isolated sites or within a network of sites linked into a common sampling design. Field procedures will likely be the same in these scenarios; however, the distribution of sampling effort in time and space and resulting appropriate analysis techniques will differ in accordance with the distribution of sites in each category and the anticipated degree of contrast in habitat condition (effect size).

To meet our objectives with available resources, we sampled streams opportunistically. However, application of these procedures within a statistically defensible sampling framework would strengthen extension of inferences beyond the sampled streams. A complete monitoring plan, regardless of specific procedures employed, that includes random sampling, known sampling probability, and spatially balanced sampling increases analysis options and robustness of conclusions (Larsen, 1997). Randomization minimizes the introduction of bias into sample selection. Variable probability designs accommodate differential sampling intensity among categories to account for different frequencies of category occurrence, while maintaining known probability of sample selection. Spatial distribution of the sample helps assure that spatially varying influences are representative (Larsen, 1997; Paulsen et al., 1998; Urquhart et al., 1998; Larsen et al., 2001).

Clearly, channel condition will respond to many geologic, climatic, vegetative, and land use influences, all of which differ spatially and temporally. Furthermore, no single response reach, even in depositional channel sections, will fully reflect channel response to upstream influences. These sources of variability contribute to the overall variance observed in response variable magnitude. Therefore large sample sizes and long time periods will be required to evaluate contrasts among land use intensity categories with reasonable statistical power. Although the duration and measurement frequency of the data set in this study are insufficient to detect trends in monitoring variables (Table 3), reductions in required monitoring duration might be possible depending on the variance structure of the indicator variables. Partitioning the total variance into components would inform careful allocation of sampling effort within and among sites to minimize the effects of these components on trend detection (Urquhart et al., 1998; Larsen et al., 2001). Roper et al. (2002) point out that among-stream variability can be reduced in three ways: (1) by stratification, e.g., on landscape characteristics, (2) by focusing more effort on fewer permanent sites, and (3) by accounting for broadly distributed sources of variability through analysis of covariance. These and other details would be part of final monitoring plan development.

Kaufmann and Larsen (2002) report relatively optimistic power analyses of EMAP data through evaluation of regional, annual, and within-season components of variation. They find that with a probability sample of 50 streams visited once per year for 12 years, they have 80% power, at $\alpha = 0.05$, to detect 2% annual trends in mean residual pool depth and percentage of sand and fines (Kaufmann, P.R., personal communication). Although these values are similar to our estimates of statistical power (Table 3), the data sets are not directly comparable. For example, EMAP protocols ignore habitat units (e.g., pools) shorter than one channel width, and residual pool depth is computed from thalweg depth measurements equally spaced at 1/3 to 1/2 channel width, rather than from direct measurement of each pool (Kaufmann et al., 1999; Kaufmann, P.R., personal communication).

Measurement error adds to the total variability of monitoring indicators, and control of these elements of precision through development and observance of strict procedures will be critical to successful effectiveness monitoring. Optimal precision would be obtained by using one team to collect channel condition data across southeastern Alaska, although this may be logistically difficult. If more crews are used, frequent and independent duplicate measurement on perhaps 20% of measured reaches would allow measurement error, and thereby data quality, to be estimated and compared to values in the literature. Conversely, if data are collected by inadequately trained individuals, these data will be subject to large measurement error, and it is less likely that they will be of sufficient quality to allow meaningful effectiveness monitoring analyses (Conquest and Ralph, 1998; Roper et al., 2002). Results from other monitoring programs demonstrate that rigorous training can reduce measurement error (Kaufmann et al., 1999; Kershner et al., 2001). During a monitoring program, frequent recalculation of power would be advisable. This allows consideration of both new data and improved approaches that reduce measurement error (Larsen et al., 2001).

One application of a long-term effectiveness monitoring program is in the evaluation of broad categories of land use practices, in the sense of adaptive management. Using the effectiveness monitoring tools provided herein, patterns of change can be examined and analyzed to inform evaluation of past actions and plan for future activities. A broadly distributed, stratified sampling design with large sample size may be the preferable approach for evaluating categories of land use practices; however, monitoring a network of individual sites over time provides valuable additional information regarding within- and among-year variability in indicators. Both approaches can be done in the context of a statistically defensible sampling design. Focusing resources on fewer sites monitored over time can facilitate collection of more detailed data regarding watershed and aquatic ecosystem condition and processes and specific land use practices affecting study reaches. Owing to small sample size (years of record) during initial monitoring, assessment of possible land use effects may rely heavily on inductive inference from examination of trends, rather than rigorous statistical testing.

Categories of land use practices can also be evaluated through a series of case studies at individual sites. Although extrapolation of results from case studies to other locations generally cannot be defended statistically, inductive reasoning can be applied to make inferences regarding similar sites. In some cases, this may be the only practical option available if access is limited and transportation is expensive; decisions based on these data may be subject to challenge on statistical grounds. Such a design might include a calibration period followed by land use treatment in one or more test watersheds, while excluding land use from a set of comparable control watersheds. Different control channels (in this case, pristine) exhibit different status and trends in channel condition variables (Figs. 6 and 8). Therefore multiple control reaches would strengthen inference derived from such a study. Following a sufficient period of data collection, which would depend on sample size, measurement frequency, and effect size, robust statistical testing for differences in temporal trends in monitoring variables and between calibration and post-treatment periods would be possible for these treatments at these sites.

The interpretive value of, and strength of conclusions drawn from, any effectiveness monitoring data set increases with the quantity and quality of information available, at the appropriate scale, on characteristics, processes, and disturbance history of the relevant watersheds and landscapes. Factors such as geology, geomorphology, hydrology, climate, soils, forest structure, disturbance processes such as mass soil movements and floods, and land use indicators such as road density are among the variables affecting channel condition. Interpretations of measured change in channel condition variables are more robust when made in the context of this watershed and landscape condition information. Such a watershed assessment framework, although not part of the procedures presented herein, strengthens the association of cause with effect (measured change in channel condition) and enhances the ability to distinguish land use related effects from variability inherent in forest channels. In depositional channels, change in a monitoring variable may imply a response to disturbance anywhere in the watershed or recovery from a previous response. Nevertheless, in many cases, this monitoring information can provide early warning to land managers, triggering further investigation that may lead to identification of, and management response to, a habitat degrading disturbance.

Variability in channel condition at all levels of land use intensity (including pristine) appears to be too large at the landscape scale to support the establishment of fixed target values for channel condition variables (see also Bisson et al., 1997 and Buffington et al., 2002b). Even in the absence of land use effects, condition of pristine channels varies with geologic, geomorphic, and climatic conditions and is subject to background disturbances that vary in magnitude and timing. Target values may be appropriate if applied at smaller scales, e.g., eco-type or channel-type subdivisions; however, varying disturbance timing, intensity, and recovery trends would still create variability in channel condition.

5. Conclusions

In this report we present an approach to effectiveness monitoring of floodplain channel condition that focuses on elements relevant to salmonid habitat in southeastern Alaska. We discuss selection of monitoring variables, demonstrate methods for collection and analyses of these data, and provide estimates of the resulting statistical power. Well-trained personnel following these procedures will be able to (1) efficiently collect field data characterizing selected effectiveness monitoring variables, (2) following a sufficient period of data collection, analyze change in channel condition, as reflected by these variables, and (3) develop conclusions regarding the relative magnitude of effects of various land use practices on channel condition. Although our objectives include neither development of a complete monitoring plan nor monitoring per se, these results provide procedures and data as a foundation for these purposes. Details of a complete, statistically defensible effectiveness monitoring plan will differ with specific information needs, anticipated future land use patterns, and availability of resources for monitoring. Regardless of these specifics, the procedures, analysis techniques, and data presented herein are valuable for use as a template and baseline for effectiveness monitoring at established locations and establishing monitoring at new locations.

Temporal and spatial variance in channel condition is large in the low-gradient, depositional channels within all land use categories examined in this study, owing to differences in watershed condition, climate, and the effects of background and anthropogenic disturbance. This makes detection of disturbance (including land use) effects difficult. An important step in enhancing change detection capability, despite this large variability, is limiting selection of monitoring variables to those that are sensitive to disturbance and can be measured objectively, precisely, and efficiently. We find that measures of channel geometry, pool density, and pool size are viable indicator variables for effectiveness monitoring. Bed surface grain size distribution is responsive to watershed disturbance, but it is difficult to measure efficiently with reasonable precision.

For contrasts among land use intensity categories, statistical power analyses indicate that 20–30 reaches per category can provide 80% power for contrasts of monitoring variables other than those involving streambed grain size distribution (Table 2). Based on power analyses of variables from six reaches, a 3% per year decrease in pool spatial density can be detected with 80% power with 7–38 years of data, depending on variance, effects size, and measurement frequency (Table 3). Adding reaches to such a design or increasing frequency of measurement at each reach generally increases power.

Our understanding of riparian and aquatic ecosystems would be improved by aquatic ecosystem monitoring in a broader range of channel types and by future research investigating aquatic productivity in these ecosystems. This study addresses floodplain type channels to examine the most integrative and sensitive stream reaches. Effectiveness monitoring of transition and headwater stream habitat is another information need important to land management agencies, in part because this includes much of the habitat of juvenile coho salmon. Reaches that are transitional from transport to depositional may, in some cases, be the first to reflect land management effects (Montgomery and Buffington, 1997). Many of the findings in this paper will be applicable to smaller and steeper channels, providing a foundation on which to build.

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References

Bathurst, J.C., 1981. Discussion of bar resistance of gravel-bed streams. J. Hydraul. Div., Am. Soc. of Civil. Eng. 104, 1587–1603.

Bauer, S.B., Ralph, S.C., 1999. Aquatic habitat indicators and their application to water quality objectives within the Clean Water Act. EPA-910-R-99-014. USEPA, Region 10, Seattle, WA.

Benda, L., Miller, D., Dunne, T., Reeves, G.H., Agee, J.K., 1998. Dynamic landscape systems. In: Naiman, R.J., Bilby, R.E. (Eds.), River Ecology and Management. Springer-Verlag, New York, pp. 261–288.

Bilby, R.E., Ward, J.W., 1991. Characteristics and function of large woody debris in streams draining old-growth, clearcut, and second-growth forests in southwestern Washington. Can. J. Fish. Aquat. Sci. 48, 2499–2508.

Bisson, P.A., Nielson, J.L., Palmason, R.A., Grove, L.E., 1981. A system of naming habitat types in small streams, with examples of habitat by salmonids during low flows. In: Armantrout, N.B. (Ed.), Proceedings of a Symposium on Acquisition and Utilization of Aquatic Habitat Inventory Information. Western Division, American Fisheries Society, Bethesda, MD, pp. 62–73.

Bisson, P.A., Reeves, G.H., Bilby, R.E., Naiman, R.J., 1997. Watershed management and Pacific salmon: desired future conditions. In: Stouder, D.J., Bisson, P.A., Naiman, R.J. (Eds.), Pacific Salmon and Their Ecosystems Status and Future Options. Chapman & Hall, New York, pp. 447–474.

Bjornn, T.C., Reiser, D.W., 1991. Habitat requirements of salmonids in streams. In: Meehan, W.R. (Ed.), Influences of Forest and Rangeland Management on Salmonid Fishes and Their Habitats Spec. Publ. 19. American Fisheries Society, Bethesda, MD, pp. 83–138.

Borenstein, M., Rothstein, H., Cohen, J., 1997. SamplePower 1.0. SPSS Inc., Chicago, IL.

Bryant, M.D., Porter, P.E., Paustian, S.J., 1991. Evaluation of a Stream channel-type system for southeast Alaska. Gen. Tech. Re PNW-GTR-267. U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station, Portland, OR, 20 pp.

Buffington, J.M., 1999. A procedure for classifying textural facies in gravel-bed rivers. Water Resour. Res. 35, 1903–1914.

Buffington, J.M., Montgomery, D.R., 1999a. Effects of hydraulic roughness on surface textures of gravel-bed rivers. Water Resour. Res. 35, 3507–3522.

Buffington, J.M., Montgomery, D.R., 1999b. Effects of sediment supply on surface textures of gravel-bed rivers. Water Resour. Res. 35, 3523–3530.

Buffington, J.M., Woodsmith, R.D., Booth, D.B., Montgomery, D.R., 2002a. Fluvial processes in Puget Sound rivers and the Pacific Northwest. In: Montgomery, D.R., Bolton, S., Booth, D.B. (Eds.), Restoration of Puget Sound Rivers. University of Washington Press, Seattle, WA, pp. 46–78.

Buffington, J.M., Lisle, T.E., Woodsmith, R.D., Hilton, S., 2002b. Controls on the size and occurrence of pools in coarse-grained forest rivers. River Res. Appl. 18, 507–531.

Bunte, K., Abt, S.R., 2001. Sampling surface and subsurface particle-size distributions in wadable gravel- and cobble-bed streams for analyses in sediment transport, hydraulics, and streambed monitoring. Gen. Tech. Re RMRS-GTR-74. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Fort Collins, CO.

Carlson, J.Y., Andrus, C.W., Froehlich, H.A., 1990. Woody debris, channel features, and macroinvertebrates of streams with logged and undisturbed riparian timber in northeastern Oregon. USA Can. J. Fish. Aquat. Sci. 47, 1103–1111.

Chamberlin, T.W., Harr, R.D., Everest, F.H., 1991. Timber harvesting, silviculture, and watershed processes. In: Meehan, W.R. (Ed.), Influences of Forest and Rangeland Management on Salmonid Fishes and Their Habitats Spec. Publ. 19. American Fisheries Society, Bethesda, MD, pp. 181–205.

Chorley, R.J., Schumm, S.A., Sugden, D.E., 1984. Geomorphology. Methuen & Co, New York, pp. 605.

Conquest, L.L., Ralph, S.C., 1998. Statistical design and analysis considerations for monitoring and assessment. In: Naiman, R.J., Bilby, R.E. (Eds.), River Ecology and Management. Springer-Verlag, New York, pp. 455–475.

Curtis, R.O., 1970. Stand density measures: an interpretation. For. Sci. 16, 403–414.

Dietrich, W.E., Kirchner, J.W., Ikeda, H., Iseya, F., 1989. Sediment supply and the development of the coarse surface layer in gravel-bedded rivers. Nature 340, 215–217.

Dingman, S.L., 1984. Fluvial Hydrology. W.H. Freeman, New York, 383 pp.

Dolloff, C.A., Hankin, D.G., Reeves, G.H., 1993. Basinwide estimation of habitat and fish populations in streams. Gen. Tech. Re SE-83. U.S. Department of Agriculture, Forest Service, Southeastern Forest Experiment Station, Asheville, NC.

Dunne, T., Leopold, L.B., 1978. Water in Environmental Planning. W.H. Freeman and Co., San Francisco, CA, 818 pp.

Dury, G.H., 1966. The concept of grade. In: Essays in Geomorphology. Heinemann, London, pp. 211–234.

Fitzpatrick, F.A., Waite, I.R., D'Arconte, P.J., Meador, M.R., Maupin, M.A., Gurtz, M.E., 1998. Revised methods for characterizing stream habitat in the National Water-Quality Assessment Program. Water Resources Investigations Report 98-4052. U.S. Geological Survey, Raleigh, N.C.

Frissell, C.A., Liss, W.J., Warren, C.E., Hurley, M.D., 1986. A hierarchical framework for stream habitat classification: viewing streams in a watershed context. Environ. Manage. 10, 199–214.

Gibbs, J.P., 1995. Monitor: Users Manual. Department of Biology, Yale University, New Haven, CT.

Gibbs, J.P., Melvin, S.M., 1997. Power to detect trends in waterbird abundance with call-response surveys. J. Wildl. Manage. 61, 1262–1267.

Gibbs, J.P., Droege, S., Eagle, P., 1998. Monitoring populations of plants and animals. BioScience 48, 935–940.

Gregory, S.V., Bisson, P.A., 1997. Degradation and loss of anadromous salmonid habitat in the Pacific Northwest. In: Stouder, D.J., Bisson, P.A., Naiman, R.J. (Eds.), Pacific Salmon and Their Ecosystems Status and Future Options. Chapman & Hall, New York, pp. 277–314.

Hack, J.T., Goodlett, J.C., 1960. Geomorphology and forest ecology of a mountain region in the central Appalachians. U.S. Geological Survey Professional Papers 347, 64.

Halupka, K.C., Bryant, M.D., Willson, M.F., Everest, F.H., 2000. Biological characteristics and population status of anadromous salmon in southeast Alaska. Gen. Tech. Re GTR-PNW-468. U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station, Portland, OR, 255 p.

Hankin, D.G., Reeves, G.H., 1988. Estimating total fish abundance and total habitat area in small streams based on visual estimation methods. Can. J. Fish. Aquat. Sci. 45, 834–844.

Harrelson, C.C., Rawlins, C.L., Potyondy, J.P., 1994. Stream channel reference sites: an illustrated guide to field technique. Gen. Tech. Re RM-245. U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station, Fort Collins, CO.

Hartman, G.F., Scrivener, J.C., 1990. Impacts of forestry practices on a coastal stream ecosystem, Carnation Creek. Brit. Columbia Can. Bull. Fish. Aquat. Sci., 223.

Heede, B.H., 1972. Influences of a forest on the hydraulic geometry of two mountain streams. Water Resour. Bull. 8, 523–530.

Hogan, D.L., 1987. The influence of large organic debris on channel recovery in the Queen Charlotte Islands, British Columbia, Canada. In: Beschta, R.L., Blinn, T., Grant, G.E., Ice, G.G., Swanson, F.J. (Eds.), Erosion and Sedimentation in the Pacific Rim. International Association of Hydrologic Sciences, pp. 343–353.

Hogan, D.L., Church, M., 1989. Hydraulic geometry in small, coastal streams: progress towards quantification of salmonid habitat. Can. J. Fish. Aquat. Sci. 46, 844–852.

Johnson, D.H., et al., 2001. Inventory and monitoring of salmon habitat in the Pacific Northwest-directory and synthesis of protocols for management/research and volunteers in Washington, Oregon, Idaho, Montana and British Columbia. Washington Department of Fish and Wildlife, Olympia, WA.

Kaufmann, P.R., Larsen, D.P., 2002. Power to detect regional trends in habitat characteristics. In: Proceedings of the Conference, Toward Ecosystem-Based Management: Breaking Down the Barriers in the Columbia River Basin and Beyond, Western Division, American Fisheries Society, Spokane, WA.

Kaufmann, P.R., Levine, P., Robison, E.G., Seeliger, C., Peck, D.V., 1999. Quantifying physical habitat in wadeable streams. EPA/620/R-99/003. EPA, Office of Research and Development, Environmental Monitoring and Assessment Program, Washington, DC.

Kaufmann, P.R., Robison, E.G., 1998. Physical habitat characterization. In: Lazorchak, J.M., Klemm, D.J., Peck, D.V. (Eds.), Environmental Monitoring and Assessment Program—Surface Waters: Field Operations and Methods for Measuring the Ecological Condition of Wadeable Streams. EPA, Washington, DC, pp. 77–118.

Keller, E.A., Swanson, F.J., 1979. Effects of large organic material on channel form and fluvial processes. Earth Surf. Process. Landforms 4, 361–380.

Kershner, J.L., et al., 2001. Effectiveness monitoring of aquatic and riparian resources in the area of PACFISH/INFISH and the biological opinions for bull trout, salmon and steelhead. Draft Plan, U.S. Department of Agriculture, Forest Service, USDI Bureau of Land Management, Logan, UT.

Larsen, D.P., 1997. Sample survey design issues for bioassessment of inland aquatic ecosystems. Hum. Ecol. Risk Assess. 3, 979–991.

Larsen, D.P., Kincaid, T.M., Jacobs, S.E., Urquhart, N.S., 2001. Designs for evaluating local and regional scale trends. BioScience 51, 1069–1078.

Leopold, L.B., Maddock, T.J., 1953. The hydraulic geometry of stream channels and some physiographic implications. US Geological Survey Professional Papers 252.

Leopold, L.B., Wolman, M.G., Miller, J.P., 1964. Fluvial Processes in Geomorphology. W.H. Freeman, San Francisco, CA.

Lisle, T.E., 1981. The recovery of aggraded stream channels at gauging stations in northern California and southern Oregon. In: Davies, R.H., Pearce, A.J. (Eds.), Erosion and Sediment Transport in Pacific Rim Steeplands Publ. No. 132. International Association of Hydrological Sciences, pp. 189–211.

Lisle, T.E., 1982. Effects of aggradation and degradation on riffle-pool morphology in natural gravel channels, northwestern California. Water Resour. Res. 18, 1643–1651.

Lisle, T.E., 1986a. Stabilization of a gravel channel by large streamside obstructions and bedrock bends, Jacoby Creek, northwestern California. Geol. Soc. Am. Bull. 97, 99–1011.

Lisle, T.E., 1986b. Effects of woody debris on anadromous salmonid habitat, Prince of Wales Island, southeast Alaska. North Am. J. Fish. Manage. 6, 538–550.

Lisle, T.E., 1987. Using "residual depths" to monitor pool depths independently of discharge. Res. Note PSW-394. U.S. Department Of Agriculture, Forest Service, Pacific Southwest Forest and Range Experiment Station, Berkeley, CA.

Lisle, T.E., Hilton, S., 1999. Fine bed material in pools of natural gravel bed channels. Water Resour. Res. 35, 1291–1304.

Lisle, T.E., Iseya, F., Ikeda, H., 1993. Response of a channel with alternate bars to a decrease in supply of mixed-size bed load: a flume experiment. Water Resour. Res. 29, 3623–3629.

Lyons, J.K., Beschta, R.L., 1983. Land use, floods and channel changes: Upper Middle Fork Willamette River Oregon (1936–1980). Water Resour. Res. 19, 463–471.

MacDonald, L.H., Smart, A.W., Wissmar, R.C., 1991. Monitoring guidelines to evaluate effects of forestry activities on streams in the Pacific Northwest and Alaska. EPA 910/9-91-001. U.S. Environmental Protection Agency, Region 10, Seattle, WA.

Meehan, W.R., 1991. Influences of Forest and Rangeland Management on Salmonid Fishes and their Habitats Spec. Publ. 19. American Fisheries Society, Bethesda, MD.

Messer, J.J., Linthurst, R.A., Overton, W.S., 1991. An EPA program for monitoring ecological status and trends. Environ. Monit. Assess. 17, 67–78.

Montgomery, D.R., Buffington, J.M., 1997. Channel-reach morphology in mountain drainage basins. Geol. Soc. Am. Bull. 109, 596–611.

Montgomery, D.R., Buffington, J.M., 1998. Channel processes, classification, and response. In: Naiman, R.J., Bilby, R.E. (Eds.), River Ecology and Management. Springer-Verlag, New York, pp. 13–42.

Montgomery, D.R., Buffington, J.M., Smith, R.D., Schmidt, K.M., Pess, G., 1995. Pool spacing in forest channels. Water Resour. Res. 31, 1097–1105.

Murphy, M.L., Heifetz, J., Johnson, S.W., Koski, K.V., Thedinga, J.F., 1986. Effects of clearcut logging with and without buffer strips on juvenile salmonids in Alaskan streams. Can. J. Fish. Aquat. Sci. 43, 1521–1533.

Naiman, R.J., Bilby, R.E., 1998. River Ecology and Management. Springer-Verlag, New York, 705 pp.

Nehlson, W., 1997. Pacific salmon status and trends–a coastwise perspective. In: Stouder, D.J., Bisson, P.A., Naiman, R.J. (Eds.), Pacific Salmon and their Ecosystems Status and Future Options. Chapman & Hall, New York, p. 685.

Neter, J., Wasserman, W., Kutner, M.H., 1983. Applied Linear Regression Models. Richard D. Irwin Inc., Homewood, IL.

Noon, B.R., Spies, T.A., Raphael, M.G., 1999. Conceptual basis for designing an effectiveness monitoring program. In: Mulder, B.S., Noon, B.R., Welsh, H.H., Reeves, G., Spies, T., Raphael, M., Palmer, C., Olsen, A.R. (Eds.), The Strategy and Design of the Effectiveness Monitoring Program for the Northwest Forest Plan. Gen. Tech. Re PNW-GTR-437. U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station, Portland, OR, pp. 21–48.

Olson-Rutz, K.M., Marlow, C.B., 1992. Analysis and interpretation of stream channel cross-sectional data. North Am. J. Fish. Manage. 12, 55–61.

Paulsen, S.G., Hughes, R.M., Larsen, D.P., 1998. Critical elements in describing and understanding our Nation's aquatic resources. J. Am. Water Resour. Assoc. 34, 995–1005.

Paustian, S.J. et al., 1992. A channel type users guide for the Tongass National Forest, southeast Alaska. Technical Paper 26, U.S. Department of Agriculture, Forest Service, Alaska Region, Juneau, AK.

Peterson, J.T., Wollrab, S.P., 1999. An analysis of potential stream fish and fish habitat monitoring procedures for the inland Northwest. BPA Report DOE/BP-25866-1. Bonneville Power Administration, Portland, OR.

Platts, W.S., Megahan, W.F., Minshall, G.W., 1983. Methods for evaluating stream, riparian, and biotic conditions. Gen. Tech. Re INT-138. U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station, Ogden, UT.

Poff, N.L., Ward, J.V., 1990. Physical habitat template of lotic systems: recovery in the context of historical pattern of spatiotemporal heterogeneity. Environ. Manage. 14, 629–646.

Poole, G.C., Frissell, C.A., Ralph, S.C., 1997. In-stream habitat unit classification: inadequacies for monitoring and some consequences for management. J. Am. Water Resour. Assoc. 33, 879–896.

Ralph, S.C., Poole, G.C., Conquest, L.L., Naiman, R.J., 1994. Stream channel morphology and woody debris in logged and unlogged basins of western Washington. Can. J. Fish. Aquat. Sci. 51, 37–51.

Reeves, G.H., Everest, F.H., Sedell, J.R., 1993. Diversity of juvenile anadromous salmonid assemblages in coastal Oregon basins with different levels of timber harvest. Trans. Am. Fish. Soc. 122, 309–317.

Reeves, G.H., Bisson, P.A., Dambacher, J.M., 1998. Fish communities. In: Naiman, R.J., Bilby, R.E. (Eds.), River Ecology and Management. Springer-Verlag, New York, pp. 200–234.

Reeves, G.H., Hohler, D.B., Larsen, D.P., Busch, D.E., Kratz, K., Reynolds, K., Stein, K.F., Atzet, T., Hays, P., Tehan, M., 2004. Effectiveness Monitoring for the Aquatic and Riparian Component of the Northwest Forest Plan: Conceptual Framework and Options. General Technical Report PNW-GTR-577. U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station, Portland, OR, 71 p.

Reiser, D.W., Bjornn, T.C., 1979. Influence of forest and rangeland management on anadromous fish habitat in western North America: habitat requirements of anadromous salmonids. Gen. Tech. Re PNW-GTR-96. U.S. Department Of Agriculture, Forest Service, Pacific Northwest Forest and Range Experiment Station, Portland, OR.

Richards, K., 1982. Rivers Form and Process in Alluvial Channels. Methuen, New York.

Roper, B.B., Scarnecchia, D.L., 1995. Observer variability in classifying habitat types in stream surveys. North Am. J. Fish. Manage. 15, 49–53.

Roper, B.B., Kershner, J.L., Archer, E., Henderson, R., Bouwes, N., 2002. An evaluation of physical stream habitat attributes used to monitor streams. J. Am. Water Resour. Assoc. 38, 1637–1646.

Rosgen, D.L., 1994. A classification of natural rivers. Catena. 22, 169–199.

Salo, E.O., Cundy, T.W. (Eds.), 1987. Streamside Management: Forestry and Fishery Interactions. Contrib. No. 57. College of Forest Resources, Institute of Forest Resources, University of Washington, Seattle, WA, 471 pp.

Schumm, S.A., 1977. The Fluvial System. Wiley, New York.

Schumm, S.A., 1985. Patterns of alluvial rivers. Annu. Rev. Earth Planet. Sci. 13, 5–27.

Shields, A., 1936. Anwendung der Aehnlichkeitsmechanik und der Turbulenzforschung auf die Geschiebebewegung. Mitteilung der Preussichen Versuchsanstalt fur Wasserbau and Schiffbau, Heft 26, 98-109. (Transl.: Ott, W.P. and van Uchelen, J.C., Publ. No. 167: Hydrodynamics Lab., California Institute of Technology, Pasadena, CA).

Smith, R.D., Sidle, R.C., Porter, P.E., Noel, J.R., 1993. Effects of experimental removal of woody debris on the channel morphology of a forest, gravel-bed stream. J. Hydrol. 152, 153–178.

Sokal, R.R., Rohlf, F.J., 1981. Biometry: The Principles and Practice of Statistics in Biological Research. W.H. Freeman and Company, New York.

SYSTAT 9 Statistics 1. SPSS Inc., Science Marketing Department.

Stouder, D.J., Bisson, P.A., Naiman, R., 1997. Where are we? resources at the brink. In: Stouer, D.J., Bisson, P.A., Naiman, R.J. (Eds.), Pacific Salmon and their Ecosystems Status and Future Options. Chapman & Hall, New York, pp. 1–10.

Sullivan, K., 1986. Hydraulics and fish habitat in relation to channel morphology. Ph.D Thesis. The Johns Hopkins University, Baltimore, MD.

Sullivan, K., Lisle, T.E., Dolloff, C.A., Grant, G.E., Reid, L.M., 1987. Stream channels: the link between forests and fishes. In: Salo, E.O., Cundy, T.W. (Eds.), Streamside Management: Forestry and Fishery Interactions Contrib. No. 57. College of Forest Resources. Institute of Forest Resources, University of Washington, Seattle, WA, pp. 39–97.

Urquhart, N.S., Paulsen, S.G., Larsen, D.P., 1998. Monitoring for policy-relevant regional trends over time. Ecol. Appl. 8, 246–257.

USDA, 1997. Tongass Land Management Plan revision, final environmental impact statement. Forest Service, Region 10 (Alaska Region), Juneau, AK.

USEPA, 1997. Environmental Monitoring and Assessment Program (EMAP) Research Strategy. EPA/620/R-98/001. U.S. Environmental Protection Agency, Office of Research and Development, Washington, DC.

Sedimentation Engineering. Manuals and Reports on Engineering Practice. American Society of Civil Engineers, New York, 745.

Wang, L., Simonson, T.D., Lyons, J., 1996. Accuracy and precision of selected stream habitat estimates. N. Am. J. Fish. Manage. 16, 340–347.

Washington Forest Practices Board [WFPB], 1993. Standard Methodology for Conducting Watershed Analysis. Version 2.0. State of Washington, Olympia, WA.

Whiting, P.J., Bradley, J.B., 1993. A process-based classification system for headwater streams. Earth Surf. Process. Landforms 18, 603–612.

Wilkinson, L., Blank, G., Gruber, C., 1996. Desktop Data Analysis with SYSTAT. Prentice-Hall, Upper Saddle River, NJ, pp. 798.

Wolman, M.G., 1954. A method of sampling coarse river-bed material. Trans. Am. Geophy. Union 35, 951–956.

Woodsmith, R.D., Buffington, J.M., 1996. Multivariate geomorphic analysis of forest streams: implications for assessment of land use impacts on channel condition. Earth Surf. Process. Landforms 21, 377–393.

Zar, J.H., 1984. Biostatistical Analysis. Prentice-Hall, Englewood Cliffs, NJ.