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ARTICLE

Detection of Regional Trends in Salmonid Habitat in Coastal Streams, Oregon

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Abstract
Freshwater habitat quality is broadly recognized as fundamental to the viability of salmonid populations. Temporal trends in freshwater habitat have rarely been quantified, however, perhaps owing to a lack of methodical and rigorous time series data sets. We present an approach for evaluating change in freshwater habitat using data from a long-term program to monitor salmonid populations and their habitats in coastal drainages of Oregon. Our goals were to (1) evaluate the presence and magnitude of an underlying linear trend in freshwater habitat condition across coastal watersheds in Oregon and (2) determine the effectiveness of the current sampling design for meeting the monitoring objectives. Four features were selected to characterize freshwater habitat: percent of pool area, large wood volume, quantity of fine sediment, and stream size. We developed a statistical model to describe the trend in these features that incorporated an error structure to account for site, year, and site-by-year variability. Spatial variability accounted for most of the overall variation, and temporal variability was minimal. Trends were detected among several of the habitat metrics and these varied by geographic region. To evaluate the efficacy of the sampling design, we generated simulated data sets with hypothesized trends of 1–2% per year and estimated trend detection power under two different survey designs, variance structures, and monitoring durations. We conclude that the power to detect trends is sensitive to the duration of the monitoring program and the structure and magnitude of the variance. The monitoring program was effective in detecting subtle trends while providing a robust data set with which to address multiple monitoring objectives. Such a monitoring program is critical to assessing the viability of salmonid populations in the Pacific Northwest and tracking recovery and conservation efforts.

A primary obstacle to the sustainability of salmonid populations is the reduction of high-quality freshwater habitats as a result of habitat destruction and fragmentation (Nickelson and Lawson 1998; Knudsen 2000). These high-quality habitats combine hydrologic and physical characteristics that determine local population persistence, often defined based on juvenile salmonid capacity, production, and survival (Nickelson and Lawson 1998). Nickelson and Lawson (1998) demonstrated that viability of coho salmon *Oncorhynchus kisutch* was dependent on the distribution and abundance of these high-quality freshwater habitats as well as conditions in the marine environment; populations with poor freshwater habitat risked extinction during periods of low ocean productivity. Both Lichatowich (1989) and Lawson (1993) speculated that coho salmon abundance, while variable and cyclical, was generally decreasing because of declining freshwater habitat quality. Accordingly, species

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recovery requires an increase in amount of high-quality habitat, or an incremental increase in quality of all habitats (Nickelson and Lawson 1998). Similar concerns have been expressed about other populations of salmon, steelhead *O. mykiss*, and trout in coastal streams (Nehlsen et al. 1991), raising issues about habitat quality from tidewater to headwaters.

Coastal streams in Oregon continue to manifest the legacies of in-channel and riparian disturbances of the past 150 years (Beschta 1979; Montgomery 2003). While some disturbances are natural (e.g., fire, flooding), influencing the diversity and complexity of habitat for salmonids through wood and coarse sediment inputs, anthropogenic disturbances often alter the natural state of in-stream habitats. These alterations not only hinder the ability of streams to support salmonids, but also lengthen the recovery time for all ecosystem components (Reeves et al. 1995). Streams have been dredged, blasted, channelized, and splash-dammed (1870–1956) to facilitate log drives, creating immediate and lasting effects on fisheries and stream channels (Miller 2010). Widespread removal of logging slash and natural debris occurred in the 1960s and early 1970s to compensate for the heavy debris load, resulting in additional simplified channels and habitats. These anthropogenic activities, coupled with natural climatic, hydrologic, and fire events, left streams simplified and in a continual state of disturbance.

In response to the legacy of habitat degradation and findings from several studies in the Alsea watershed (Hall et al. 1987; Hall and Lantz 1969), land use restrictions were put into practice (Hariston-Strang et al. 2008) and restoration efforts were expanded during recent decades to conserve declining fish populations. Additional motivation for state and federal agencies to act resulted from a regional assessment identifying more than 200 stocks of Pacific salmon and steelhead at risk of extinction or of concern in the Pacific Northwest (Nehlsen et al. 1991). Despite such measures, many Pacific salmon populations have continued to decline, and 28 stocks have been added to the federal list of threatened or endangered species (http://www.nwr.noaa.gov/ESA-Salmon-Listings/upload/snapshot-7–09.pdf). Broad-scale monitoring was necessary to track these efforts and evaluate the status and trends in aquatic habitat as the new protection standards and restoration strategies were implemented.

Salmonid populations can be resilient to subtle changes that occur, often by adapting their life history patterns, reproductive rates, and dispersal modes (Reeves et al. 1995; Bisson et al. 2009). However, salmonid population responses are difficult to detect because of the variation in environmental processes and biological factors (density-dependence, incidence of predators, disease and parasites) that directly regulate abundance (Milner et al. 2003). Further, fluctuating ocean conditions driven by decadal and annual climatic patterns not only strongly influence the temporal variability seen in salmonid abundances but may mask the long-term trends in freshwater habitat quality (Lawson 1993). Monitoring of habitats and populations provides a context from which to assess current status and determine temporal trends in aquatic resource responses to natural and anthropogenic variability.

Rigorous monitoring designs are necessary to account for the intrinsic temporal and spatial variability inherent in ecological data (Oakley et al. 2003; Lovett et al. 2007). Although limited, much of the previous work to evaluate change in freshwater habitat has been site-specific and often related to restoration effectiveness monitoring at that scale (Roni et al. 2002; Larsen et al. 2004; Klein et al. 2007). There are only a few programs in place to adequately address regional trends in freshwater habitat (i.e. U.S. Forest Service [USFS] Aquatic and Riparian Effectiveness Monitoring Program; Environmental Protection Agency Environmental Monitoring and Assessment Program [Urquhart et al. 1998]; USFS and Bureau of Land Management PACFISH/INFISH Biological Opinion Effectiveness Monitoring Program [Dugaw et al. 2005]). Larsen et al. (2004) suggested that addressing trends in habitat conditions at a regional scale may be most beneficial because habitat degradation has occurred at these extents over decades. Further, salmon recovery efforts often consider issues occurring at these broad scales in response to regional disturbances affecting salmon populations.

Gradual habitat change over broad regions is often difficult to detect in the short term, and it may require several years to discern consistent trends. The ability to detect temporal trends is dependent on not only the survey design but also the duration of sampling and the magnitude of the spatial and temporal variability (Urquhart et al. 1993; Stow et al. 1998; Larsen et al. 2004). Correspondingly, the probability of detection, also sensitive to the sampling design upon which the data are based, relies on the consideration of additional factors (e.g., desired level of precision, underlying statistical model, variance, sample size; Seavy and Reynolds 2007). A realistic appraisal of the probability of detection, or statistical power, prompts investigators to consider biological significance in addition to statistical significance (Van Strien et al. 1997; Hatch 2003). Statistical power has also been used to judge whether nonsignificant results in trend analysis can be interpreted with confidence (Bryant et al. 2004) and to define and assess monitoring goals and evaluate the effectiveness of ongoing efforts (Van Strein et al. 1997; Hatch 2003; Seavy and Reynolds 2007). Nonsignificant trend results may in fact be very important relative to long term negative trends in habitat condition, particularly in relation to the viability of fish populations.

In 1997, the Oregon Department of Fish and Wildlife (ODFW) established a long-term sampling program in Oregon coastal watersheds to monitor the status and trends of salmonid populations and habitats and to evaluate their responses to conservation and restoration efforts throughout the state (Oregon Coastal Salmon Restoration Initiative 1997). A spatially and temporally explicit sampling design was established to balance the need for information about both status and trend (Stevens and Olsen 2004). The design included repeated annual sampling at some sites to achieve power sufficient for trend detection and added new sites each year to increase habitat coverall and optimize status estimates (Firman and Jacobs 2001; Jones et al. 2001). A more conventional design based on an annual resample of randomly selected sites would have maximized the ability to detect trends but would have been less suited to represent status across the study area. The monitoring program established a consistent protocol and sampling design to quantify the condition of freshwater habitats and estimate the abundance of juvenile and adult coho salmon (Firman and Jacobs 2001; Jones et al. 2001). While the fish sampling focused on coho salmon, aquatic habitats were surveyed throughout coastal stream networks that support salmon, steelhead, and anadromous or resident cutthroat trout *O. clarkii*.

We examined whether stream habitat in coastal basins of Oregon has stabilized during the past decade. Our objectives were to (1) evaluate the presence and magnitude of an underlying linear trend in aquatic habitat across coastal watersheds in Oregon within a 10-year period (1998–2007) and (2) determine whether the sampling design is effective in evaluating linear trends under current and alternative hypothetical scenarios. We followed the logic of Urquhart et al. (1993) that any patterned, consistent change over time will have an underlying linear component, whether or not the trend is in fact linear. This approach has been widely applied to physical surveys of stream and lake habitats (Larsen et al. 2001; Larsen et al. 2004) and biotic surveys of fish (largemouth bass *Micropterus salmoides*, walleye *Sander vitreus*; Wagner et al. 2007; Koslow et al. 2002), reptiles (American alligator *Alligator mississippiensis*; Nikerson and Brunell 1997), butterflies (Van Strein et al. 1997), mollusks (Family: Sphaeriidae; Gray and Burlew 2007), and Artic-alpine origin vegetation (Lesica and Steele 1996). We modified a model originally proposed by VanLeeuwen et al. (1996) and applied by Larsen et al. (2001) and Piepho and Ogutu (2002). Although the model we present is not novel, the data set on which it is based is unprecedented. The results presented here represent a unique effort to quantify a trend in freshwater fish habitat in a major ecological province of the Pacific Northwest and assess the utility of the sampling design at a regional scale.

METHODS

Study Area

Our analysis included all wadeable streams in coastal catchments larger than 0.6 km^2 west of the Cascade Mountains and south of the Columbia River in Oregon (Figure 1). This area was divided into five gene conservation units or monitoring strata—North Coast, Midcoast, Mid-South Coast, Umpqua, and South Coast—based on studies of coho salmon genetic variation and life history traits (Kostow 1995; Figure 1). These boundaries are further demarcated by geologic characteristics. The monitoring strata represent meaningful biological strata and the regional scale at which policy and recovery planning occurs for coho salmon in Oregon. The Coast Range of Oregon is characterized by diverse environmental patterns in landform, geology, and climate (Spies et al. 2002). The region is primarily composed of marine sandstone and shale or basaltic lithology in the four northern strata and a complex mix of granitic, sedimentary, metamorphic, and extrusive rock in the South Coast stratum (Spies et al. 2002). The moderate, maritime climate results in mild, wet winters and dry summers, but relatively narrow seasonal fluctuations in temperature. Most of the precipitation falls as rain. Peak flows occur between November and March, and base stream flows occurring between July and October (Spies et al. 2002). The coastal basins contain important habitat for five anadromous salmonids: coho salmon, steelhead, coastal cutthroat trout *O. clarkii clarkii*, Chinook salmon *O. tshawytscha*, and chum salmon *O. keta*.

Survey Design

Sample sites were selected within each monitoring stratum from a 1:100,000-scale hydrography layer (1998–2006) developed by the U.S. Geological Survey (USGS) and a 1:24,000-scale hydrography layer (2007) modified by ODFW. Potential sample sites for a given year were selected within each monitoring stratum using a generalized random-tessellation stratified design. This design provides a random selection of sites that are evenly dispersed over the hydrography extent (i.e., spatially balanced random sample; Stevens and Olsen 2004). Each monitoring stratum had a target sample size of 45 sites per year. A temporal component was further imposed using sampling intervals based on a 27-year rotating-panel design that included four temporal strata: (1) a single annual panel visited once each year, (2) three 3-year panels visited alternately once every 3 years, (3) nine 9-year panels visited alternately once every 9 years, and (4) twenty-seven 27-year panels effectively visited once only (Stevens and Olsen 2004). The 3-year interval underlying the panel structure was chosen to coincide with the duration of the typical coho salmon life cycle; however, the 3-year habitat sites extended beyond the distribution of coho salmon. All sites where at least two surveys were conducted within the 10-year period (1998–2007) were included in this analysis. Unfortunately, no annual or 3-year site surveys were conducted in 2004 because of logistical constraints; therefore, this year was excluded from the analysis. Approximately 10% of the sites within each monitoring strata were re-surveyed each year to assess within-year variability and to compare the precision of estimates among field crews. These sites were chosen at random from all the sites surveyed each year (regardless of the panel to which a site was assigned). A total of 984 observations from 237 unique sites across the five monitoring strata were included in the subsequent analyses.

Stream Habitat Features

Aquatic habitat surveys were conducted from mid-June through late September each year. Surveys followed the methods of Moore et al. (2007) and described channel and valley morphology, instream habitats, and riparian areas. Survey reach lengths were approximately 500 or 1,000 m depending on stream size. These survey lengths enabled delineation of 20–40 habitat

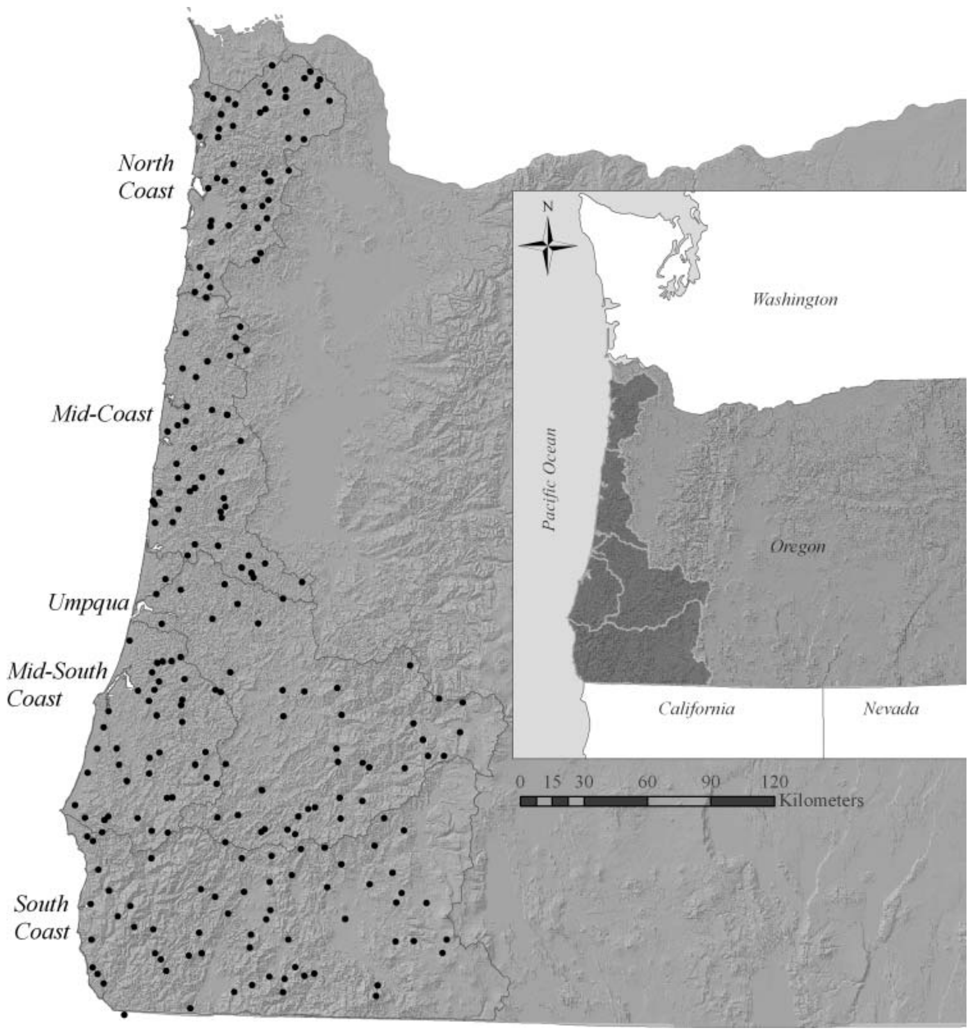

FIGURE 1. Survey site locations within monitoring strata of coastal Oregon (North Coast, Midcoast, Mid-South Coast, Umpqua, and South Coast) used to evaluate freshwater habitat condition across the study region.

units at a site, providing adequate detail to characterize certain patchy habitat features (Jones et al. 2001).

We assessed trends in four habitat features: percent of pool area, large wood volume, quantity of fine sediment, and active channel width (Table 1). Selection of these features was based on the ability to meet statistical assumptions as well as their biological pertinence. When distinguishing these features from similar habitat metrics summarized from field data, we assessed

TABLE 1. In-stream habitat features and units of measure used to evaluate coastal Oregon streams. All features were summarized at the reach scale (500- or 1,000-m reach lengths).

Habitat feature	Unit	Data transformation	Description
Pool habitat	Percent		Area of scour and dammed pools as a percentage of total wetted channel area
Fine sediment	Percent		Percentage of streambed area classified as sand, silt, and organic substrates (<2 mm)
Wood volume	m^3/100 m	Log_e	Volume of wood per 100 m of primary channel length
Active channel width	m	Log_e	Distance across stream at bank-full flow; bank-full levels are attained on average every 1.5 years.

their overall data distribution, influence by watershed processes, and representativeness of salmonid habitat. These four features characterize stream morphology, roughness, and size contributing, in one way or another, to the capacity and quality of instream habitat for salmonids (Bjornn and Reiser 1991; Rosenfeld et al. 2000).

The distribution and abundance of juvenile salmonids is largely influenced by the availability and amount of pool area. Salmonids use pools differently depending on the species, life stage, and time of year (Bjornn and Reiser 1991). Pools are slow, deep habitats that provide suitable water velocities and cover from predators for juveniles and spawning gravel in the tail-outs for adults. Juvenile salmonid densities are often higher in these habitats (Rosenfeld et al. 2000). In this study, pool area was calculated as the percent by area of scour and dammed pools within the wetted channel of a reach.

Large wood is a frequent component of high-quality instream habitats, exerting influence on many instream habitat features as well as affecting channel processes and patterns (Montgomery 2003). Although the role of large wood varies with stream size, instream wood provides structural complexity and habitat heterogeneity (Reeves et al. 1995). Large wood is used in habitat restoration projects to increase salmonid abundance and survival by creating refuge habitat. This habitat feature alters flow dynamics, creates velocity breaks and scours pools, and forms secondary channel habitats (Bisson et al. 1987; Cederholm et al. 1997; Johnson et al. 2005). Instream wood retains gravel and sediments and provides nutrients, refuge, and cover for freshwater biota (Bjornn and Reiser 1991). Wood volume was calculated as the total wood volume (i.e., length$\cdot\pi\cdot$radius^2) per 100 m of surveyed stream within a reach.

The composition of in-stream sediment is a useful measure to convey the extent of disturbance and erosion in a drainage (Bjornn and Reiser 1991). The degree of sedimentation is also dictated by immutable factors such as stream gradient and the underlying geologic template. High quantities of fine sediments can indicate habitat homogeneity and poor water quality and can influence permeability and embeddedness of channel substrate (Kaufmann and Hughes 2006). Fine sediment also reduces the abundance of aquatic invertebrates, which are important prey for salmonids (Suttle et al. 2004). Moreover, high sediment loads can affect proximal water exchange and O_2 availability, directly influencing salmonid incubation and reducing the survival of embryos (Bjornn and Reiser 1991). The quantity of fine sediment was calculated as the percent of sediments less than 2 mm in diameter, visually estimated in each habitat unit (e.g., pools, riffles, rapids, and cascades) and averaged across a reach.

The extent of the active channel (active channel width, ACW) defines the margins of a stream channel. These margins represent the height and width of flow events that occur on average every 1.5 years. A common metric, ACW is obtained by many stream habitat protocols (Kaufmann et al. 1999; Reeves et al. 2004) and is often used as a predictor for species presence because it correlates with stream size and geographic placement within a watershed (Rosenfeld et al. 2000).

Variance Partitioning and Trend Model

To estimate linear trends over 10 years (1998–2007) for each of the four habitat features (response variables), we modified a model proposed by VanLeeuwen et al. (1996). For our purposes, we define trend as a directional change over time in a habitat feature. We used site specific trends to draw inferences about trends at regional scales. For a variety of reasons (e.g., inability to access private lands, forest fires, streams that were dry in drought years, and time constraints) the number of sites surveyed within the period varied among the five monitoring strata, as did the selection and number of between-year and within-year site revisits. Thus, the design was unbalanced. We represented the response variable (e.g., wood volume) measured at the *k*th revisit to site *j* in monitoring stratum *h* during year *i* as

$$z_{hijk} = \alpha_h + \beta_h t + y_i + s_{j(h)} + (s\cdot y)_{ij(h)} + e_{hijk};$$

t	=	the time in years;
α_h	=	the monitoring stratum intercept parameter;
β_h	=	the monitoring stratum linear trend (slope) parameter;
y_i	=	random year component;
$s_{j(h)}$	=	site component;
$(s\cdot y)_{ij(h)}$	=	site-by-year interaction component; and
e_{hijk}	=	residual error component.

Thus, this formulation models the habitat response as a monitoring stratum specific linear function of time, variance being determined by four independent normally distributed random effects attributable to year (coherent temporal) variability ($\sigma_y{}^2$), inherent site variability ($\sigma_s{}^2$), site-by-year interaction variability ($\sigma_{sy}{}^2$), and residual error variability ($\sigma_e{}^2$). We used restricted maximum likelihood to estimate the variance components and based all hypothesis tests on the type III test of fixed effects with the Kenward–Rogers method (Kenward and Rogers 1997) to estimate the degrees of freedom for the denominator (Littell et al. 2006). The linear mixed model was fit using Proc Mixed in SAS (SAS 2000; Littell et al. 2006).

We performed separate analyses for each of the four response variables: percent of pool area, quantity of fine sediments, wood volume (m^3/100 m), and active channel width (m). Wood volume and active channel width were transformed using the natural logarithm to better approximate the assumptions of the linear model. In each case we initially considered the possibility of a heterogeneous covariance structure allowing variance components to differ across monitoring strata. We were able to eliminate this model in favor of the simpler, homogenous model when these models were assessed using likelihood-based methods of model selection (Akaike information criterion [AIC]; Burnham and Anderson 2002). The model with the lower AIC was deemed more favorable. As outlined in Littell et al. (2006),

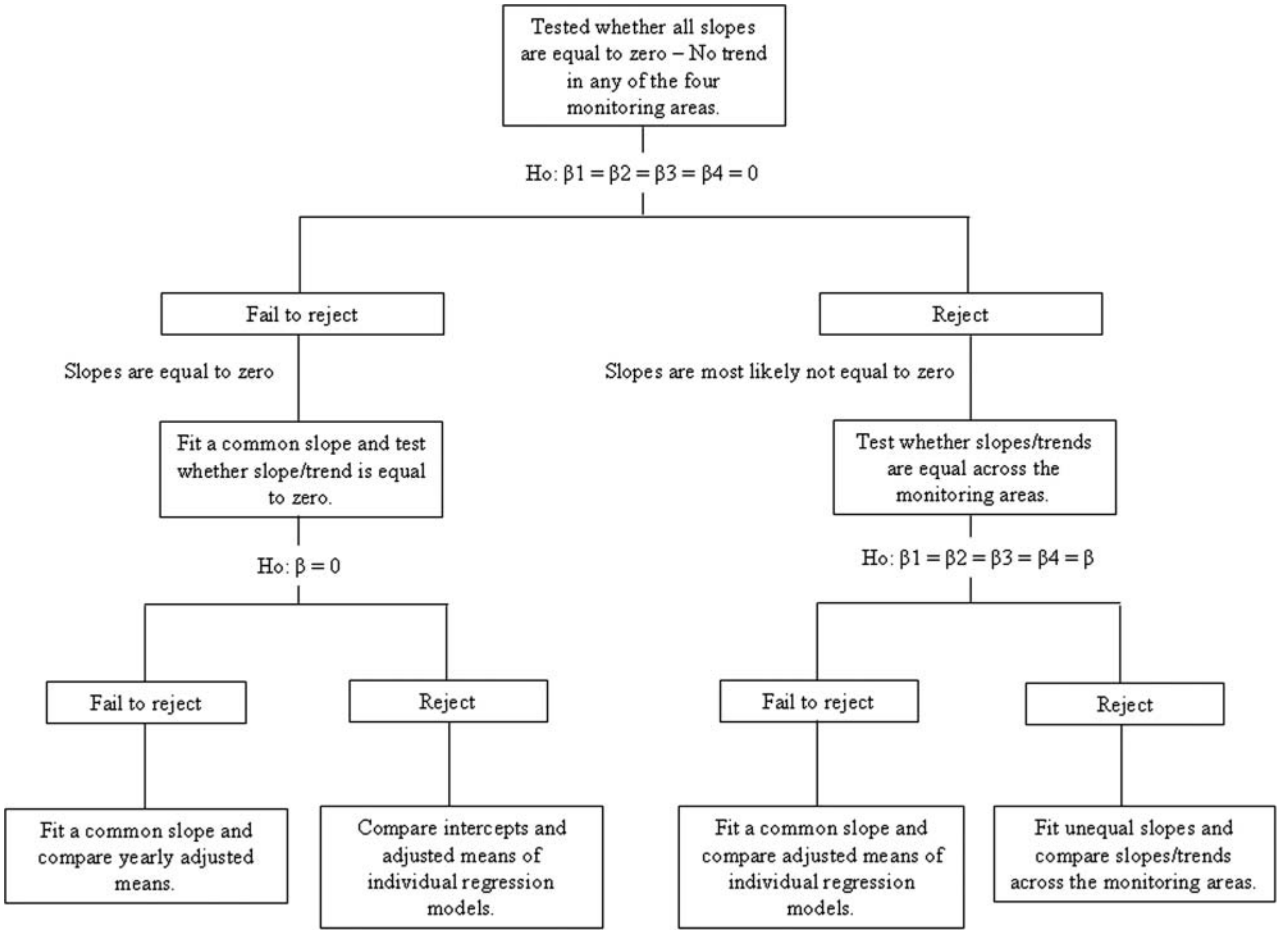

FIGURE 2. Flow diagram of the hypothesis testing and modeling process used to evaluate freshwater habitat condition in coastal Oregon. Separate analyses were completed for each habitat feature; β represents the monitoring stratum linear trend (slope) parameter.

we tested a procession of hypotheses in a procedure represented in Figure 2. The restricted maximum likelihood estimates of the four variance components—σ_y^2, σ_s^2, σ_{sy}^2, and σ_e^2—were used to understand how the variability in the system is partitioned and also as a basis for subsequent simulations.

Simulations and Power Analysis

Data sets.—To investigate the efficacy of the rotating-panel survey design for detecting trends and to compare it with a more conventional single-panel design in which every site is visited at least once in each year, we generated simulated data sets based on the postulated model as described above (see *Variance Partitioning and Trend Model*) and each survey design, testing the null hypothesis of no linear trend under assumed linear trends of 0, 1, and 2%. Our goal was to approximate the annual intended sampling effort for three different monitoring periods: 5, 10, and 15 years. Owing to the size and complexity of the data sets and associated computational expense, we limited consideration to a single monitoring stratum. Each data set simulated visits to 48 sites, plus an additional 5 sites (about 10%) randomly chosen within-year revisits, making 53 observations in any 1 year. For the conventional design, 48 distinct sites are visited over the course of any monitoring period because each site is visited each year. Under the panel design, the 48 sites visited in any 1 year come equally from four panels (12 sites per panel) according to the panel rotation schedule. Thus, over a 10-year monitoring period, for example, exactly 276 distinct sites are visited ($23 \times 12 = 276$). These consist of 1×12 annual-panel sites, $3\times$ 12 3-year-panel sites, 9×12 9-year-panel sites, and 10×12 27-year-panel sites. Recall that annual-panel sites are visited each year, while 3-year-panel and 9-year-panel sites are revisited every 3 years and every 9 years. In a 10-year period, only the 12 sites of the first 9-year panel are revisited, which occurs in year 10. None of the 12 sites of any of the ten 27-year panels are revisited in the 10 years.

Analysis.—For each trend, monitoring period, and survey design scenario (Table 2), we used estimates from the wood volume data set as the variance component parameters σ_y^2, σ_s^2, σ_{sy}^2, and σ_e^2 and the average of the estimated intercepts (estimated means for the monitoring stratum at the start of the

TABLE 2. Scenarios used to estimate statistical power in a study of habitat in coastal Oregon streams. Simulated data sets were based on estimates from the wood volume data set from a single monitoring stratum.

Monitoring period (years)	Yearly Linear trend (%)	Sampling design
5	0	Panel design
10	1	Panel design
15	2	Panel design
5	0	Conventional design
10	1	Conventional design
15	2	Conventional design
5	0	Panel design with increase in year variance
10	1	Panel design with increase in year variance
15	2	Panel design with increase in year variance
5	0	Conventional design with increase in year variance
10	1	Conventional design with increase in year variance
15	2	Conventional design with increase in year variance

monitoring period) from the same data as the initial population mean parameter μ. With only one monitoring stratum, the simulated random response z_{ijk} at revisit k to site j during year I, given hypothesized trend $\beta = 0$, 0.01μ or 0.02μ, was thus generated as

$$z_{ijk} = \mu + \beta t + U_i + U_j + U_{ij} + U_{ijk},$$

where $t = i$ is the time in years and U_i, U_j, U_{ij}, and U_{ijk} are simulated independent random normal variates with mean zero and respective variances $\sigma_y{}^2$, $\sigma_s{}^2$, $\sigma_{sy}{}^2$, and $\sigma_e{}^2$. We used parameter estimates from wood volume because it is representative of the other features with respect to the variance structure and because the importance of this attribute is universally acknowledged within stream ecology. Given that the variance structures of other habitat features were similar to wood volume, it is likely that similar power would be achieved for other variables under comparable monitoring scenarios.

We evaluated the statistical power of the test as the as the probability of rejecting the null hypothesis when it is false (calculated as 1 − type II error) and estimated it as the proportion of significant tests of the hypothesis H_0: $\beta = 0$ of no linear trend over 1,000 independent simulated data sets at $\alpha = 0.10$ (as used previously by Larsen et al. 2001 to illustrate trend detection power). Inclusion of the zero-trend scenario data sets allowed estimation of the probability of rejecting the null hypothesis when it is true (type I error) as the proportion of significant tests given an actual trend of zero, again on the basis of 1,000 independent simulated (zero-trend) data sets and at $\alpha = 0.10$. Standard errors for estimates of both statistical test power and the type I error rate were calculated as $\sqrt{\frac{p(1-p)}{1000}}$, where p is the observed proportion of significant test results in each set of 1,000 simulations. The computation expense of the simulations forced the use of the SAS default "containment" method for determining the denominator degrees of freedom rather than a presumably more appropriate method, such as Kenward-Rogers or Satterthwaite (Satterthwaite 1946; Kenward and Rogers 1997).

Finally, to investigate how greater year-to-year variability might impact trend detection, we performed another set of 1,000 test simulations, including type I error estimation, supposing a 10-fold increase in year-to-year variability but reducing site variability to keep the total variability constant. Through this exercise, we assessed the effect of an alternative partitioning of the variance.

RESULTS

Variance Partitioning

The relative proportions of total variance attributed to each component were similar among all of the habitat features. Site variation made up 72–88% of the total variability in the data (Table 3; Figure 3). Coherent temporal variability was minimal compared with the spatial variability for all four variables, ranging from 0.18% to 0.75% for the year component and 3% to 15% for the site-by-year interaction component. The remaining residual variation ranged from 6.7% to 11.6%. The ACW feature had the highest proportion of residual error. The pool habitat feature had the highest proportion of the total variance associated with the year and site-by-year interaction components.

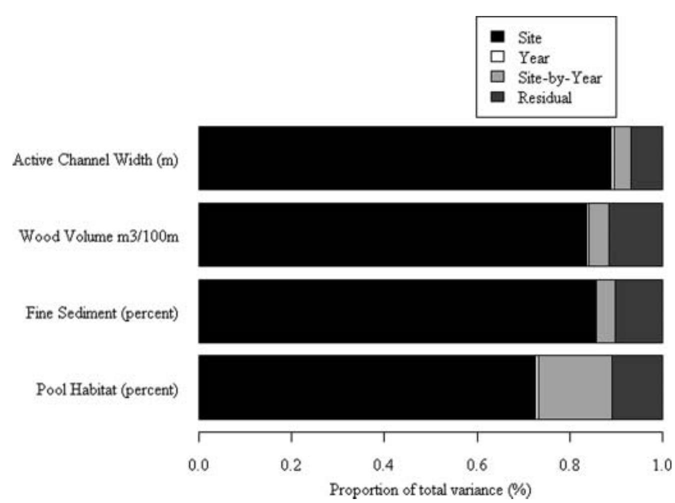

FIGURE 3. Relative proportion of variation contributed by each random variance component for the four stream habitat features evaluated in the trend and power analyses of coastal Oregon stream habitat.

TABLE 3. The proportions of variance, relative to the total variance, attributed to each of the four variance components for each of the habitat features examined in coastal Oregon streams.

	Variance components			
Habitat feature	Site	Year	Site × Year	Residual
Pool area (%)	72.54	0.75	15.70	11.01
Fine sediment (%)	85.51	0.18	4.02	10.29
Wood volume ($\log_e$[m^3/100 m])	83.62	0.52	4.17	11.69
Active channel width ($\log_e$ [m])	88.90	0.51	3.81	6.78

Trend Model

Linear trends were detected in three of the four habitat features. We rejected the global hypothesis H_0: $\beta_1 = \beta_2 = \beta_3 = \beta_4 = \beta_5 = 0$ of no trend in any of the five monitoring strata for wood volume, percent of fine sediments, and active channel width. We fit a common-slope model to the pool habitat feature and compared the least-squares means among the monitoring strata (Tables 4–6). Pool habitat percentages did vary significantly among monitoring strata. The South Coast and Umpqua strata had the lowest values and were significantly lower than the northern most monitoring strata (Tables 4, 5).

For wood volume, fine sediment, and ACW features, we then tested the hypothesis H_0: $\beta_1 = \beta_2 = \beta_3 = \beta_4 = \beta_5 = \beta$ of equal trends across all monitoring strata (Table 4). We rejected this hypothesis for each of the variables and fit an unequal slopes model. For wood volume, a linear decrease of −0.05 $\log_e$(m^3/100 m) per year for the 10-year monitoring period was detected in the North Coast region (Tables 4, 5), and ACW in the South Coast showed an estimated increase of 0.03 $\log_e$(m/year). Note that trend estimates on the $\log_e$ scale translate directly into estimated multiplicative as opposed to additive effects in terms of the original units, so that these effects correspond to estimated changes of approximately −4.9% per year in North Coast wood volume and 3% per year in South Coast active channel width. A negative estimated trend or a decrease of −1.8% fine sediments per year was detected in the North Coast, and a positive estimated trend or an increase of 1.0% of fine sediments per year was detected in the Mid-South Coast (Tables 4, 5). Sixteen of the 20 variable–monitoring strata combinations did not show a significant trend over the 10-year period. No discernable change was detected in pool habitat area, wood volume, fine sediment, or ACW in most strata.

Simulations and Power Analysis

To estimate power with reasonable precision, we used 1,000 simulated data sets for each survey design–trend combination.

TABLE 4. Model output with overall slope estimates addressing hypothesis 1 ($H_{0,1}$: $\beta_1 = \beta_2 = \beta_3 = \beta_4 = \beta_5 = 0$; i.e., no trend in any of the five monitoring strata) and hypothesis 2 ($H_{0,2}$: $\beta_1 = \beta_2 = \beta_3 = \beta_4 = \beta_5 = \beta$; i.e., equal trends across all monitoring strata), associated effect, F-value, p-value ($\alpha = 0.05$), and the result of the test; MA = monitoring stratum.

Habitat feature	Hypothesis	Effect	F-value	p-value ($\alpha = 0.05$)	Test result	implication
Pool area (%)	$H_{0,1}$	Time × MA	1.21	0.3126	Fail to reject	Fit a common slope model
Fine Sediment (%)	$H_{0,1}$	Time × MA	9.97	0.0001	Reject	Test $H_{0,2}$: Slopes are equal
	$H_{0,2}$	Time	0.31	0.5915		
		Time × MA	12.72	0.0001	Reject	Fit an unequal slope model
Wood volume equal ($\log_e$ [m^3/100 m])	$H_{0,1}$	Time × MA	3.05	0.0152	Reject	Test $H_{0,2}$: Slopes are equal
	$H_{0,2}$	Time	1.81	0.2207		
		Time × MA	3.51	0.0077	Reject	Fit an unequal slope model
Active channel equal width ($\log_e$ [m])	$H_{0,1}$	Time × MA	4.60	0.0012	Reject	Test $H_{0,2}$: Slopes are equal
	$H_{0,2}$	Time	0.31	0.5978		
		Time × MA	5.63	0.0002	Reject	Fit an unequal slope model

TABLE 5. Model output based on hypothesis 1 ($\beta_1 = \beta_2 = \beta_3 = \beta_4 = \beta_5 = 0$) for pool habitat and hypothesis 2 ($\beta_1 = \beta_2 = \beta_3 = \beta_4 = \beta_5 = \beta$) for the remaining habitat features. The *p*-values ($\alpha = 0.05$) are based on intercept and slope estimates for the fixed effects of time and time × ma.

		Intercept		Slope	
Habitat feature	Monitoring strata	Value (SD)	p-value	Value (SD)	*p*-value
Pool area (%)	North Coast	37.66 (3.6)	<0.001	0.39 (0.47)	0.406
	Midcoast	36.26 (3.8)	<0.001	0.87 (0.48)	0.078
	Mid-South	32.77 (3.9)	<0.001	0.65 (0.51)	0.213
	Umpqua	24.51 (3.8)	<0.001	−0.24 (0.50)	0.632
	South Coast	17.85 (3.2)	<0.001	−0.02 (0.40)	0.965
Fine sediment (%)	North Coast	38.53 (3.4)	<0.001	−1.86 (0.35)	0.001
	Midcoast	29.72 (3.7)	<0.001	−0.32 (0.37)	0.387
	Mid-South	27.86 (3.7)	<0.001	1.06 (0.39)	0.006
	Umpqua	27.17 (3.7)	<0.001	−0.39 (0.38)	0.301
	South Coast	23.89 (3.0)	<0.001	0.20 (0.25)	0.439
Wood volume ($\log_e$ [m^3/100m])	North Coast	3.12 (0.20)	<0.001	−0.05 (0.02)	0.013
	Midcoast	2.55 (0.21)	<0.001	0.03 (0.02)	0.096
	Mid-South	2.58 (0.22)	<0.001	−0.01 (0.02)	0.595
	Umpqua	2.43 (0.21)	<0.001	−0.01 (0.02)	0.681
	South Coast	2.10 (0.18)	<0.001	−0.01 (0.01)	0.506
Active channel width ($\log_e$ [m])	North Coast	1.98 (0.10)	<0.001	−0.007 (0.01)	0.467
	Midcoast	1.89 (0.11)	<0.001	−0.01 (0.01)	0.220
	Mid-South	1.71 (0.11)	<0.001	0.01 (0.01)	0.197
	Umpqua	1.41 (0.11)	<0.001	0.01 (0.01)	0.292
	South Coast	1.70 (0.09)	<0.001	−0.03 (0.01)	0.011

We used the variance components estimates obtained from analysis of the wood volume data with site = 1.58, year = 0.01, site × year = 0.08, and residual = 0.22 (total variance = 1.89). The intercept term, averaged over the five estimated intercepts from the wood volume model, was 2.6.

Under both the panel and conventional sampling designs, the probability of detecting a trend in the response variable increased with monitoring period (5, 10, or 15 years) and with the size of the hypothesized trend (1% or 2%; Table 7). Associated type I error rates under the panel design were statistically indistinguishable from the nominal α level (0.10) for the 5-year and 15-year periods but were evidently slightly inflated for 10-year period. Under the conventional design, on the other hand, the type I error rates were greater than the nominal α level for each of the monitoring periods, somewhat undermining test reliability under the corresponding trend scenarios (Table 7).

TABLE 6. Differences in least-squares means among monitoring strata for the pool habitat feature in five regions of coastal Oregon: North Coast (NC), Midcoast (MC), Mid-South (MS), Umpqua (UMP), and South Coast (SC). The *p*-value is based on the adjustment using Tukey–Kramer in SAS.

Strata	Estimate (SE)	*t*-value	*p*-value
NC–UMP	15.61 (4.50)	3.47	0.006
NC–SC	21.64 (4.09)	5.22	<0.001
MC–UMP	16.07 (4.66)	3.45	0.006
MC–SC	21.87 (4.27)	5.12	<0.001
MS–UMP	12.86 (4.62)	2.78	0.047
MS–SC	17.50 (4.28)	4.09	0.000

An increase in the year variance component reduced the ability to detect trend. With a 10-fold increase in year-to-year variability and with total variation held constant, simulation variance components were as follows: site = 1.49, year = 0.10, site × year = 0.08, and residual = 0.22 (total variance = 1.89). Trend detection probability under the panel design again increased with the length of the monitoring period. However, the power was generally less than the smaller year-to-year variability scenario, especially at the longer monitoring periods (Table 7). Estimates of the type I error rate under the panel design in this instance were all very close to the nominal α level (0.10). Trend detection probability under the conventional design was likewise reduced when year-to-year variability accounted for a larger proportion of the total, and this reduction was more pronounced for longer monitoring periods. Additionally, and somewhat paradoxically, corresponding type I error rates all were slightly to moderately inflated.

TABLE 7. Power analysis based on 1,000 independent simulated data sets with linear trend detection rates (means + SEs) employing the panel and conventional survey designs for monitoring periods of 5, 10, and 15 years. Two sets of simulations were done. In the first set, the variance components were as follows: site = 1.58, year = 0.01, site × year = 0.08, and residual = 0.22 (total variance = 1.89); the intercept = 2.6. In the second set, we assumed a 10-fold increase in the year variance (year = 0.10) and adjusted the site variance downward (site = 1.49) so that the total variance would remain the same. The SAS type III test of fixed effects was used, with the default (containment) method for computing the degrees of freedom in the denominator and an α of 0.10.

		Yearly Linear Trend		
Monitoring effort (years)	Design	0%	1%	2%
			First set of simulations	
5	Panel	0.098 ± 0.009	0.153 ± 0.011	0.282 ± 0.014
10	Panel	0.147 ± 0.011	0.512 ± 0.016	0.981 ± 0.004
15	Panel	0.098 ± 0.009	0.937 ± 0.008	1.000 ± 0.000
5	Conventional	0.133 ± 0.011	0.224 ± 0.013	0.400 ± 0.015
10	Conventional	0.122 ± 0.010	0.601 ± 0.015	0.978 ± 0.005
15	Conventional	0.143 ± 0.011	0.955 ± 0.007	1.000 ± 0.000
			Second set of simulations	
5	Panel	0.122 ± 0.010	0.116 ± 0.010	0.312 ± 0.015
10	Panel	0.082 ± 0.009	0.286 ± 0.014	0.488 ± 0.016
15	Panel	0.107 ± 0.010	0.364 ± 0.015	0.783 ± 0.013
5	Conventional	0.170 ± 0.012	0.212 ± 0.013	0.231 ± 0.013
10	Conventional	0.140 ± 0.011	0.215 ± 0.013	0.450 ± 0.016
15	Conventional	0.122 ± 0.010	0.412 ± 0.016	0.839 ± 0.012

Finally, we note that the statistical power under the conventional design significantly exceeds that of the panel design in five of the twelve scenarios, though it is statistically indistinguishable in five other scenarios. It should be observed, however, that this assessment makes no attempt to adjust for possible differences in the type I error rate under the two designs for a given scenario (Table 7).

DISCUSSION

We had a unique opportunity to assess trends in freshwater habitat, benefiting from a long-term data set founded on rigorous design principles. The monitoring design has shown itself to be useful for describing the status of aquatic habitat (Jones et al. 2001; Rodgers et al. 2005; Anlauf et al. 2009), and this study indicates that it holds promise for detecting subtle trends in aquatic habitats important to salmonid productivity. The statistical model we chose emphasized quantification of error variance and sampling effort with respect to repeat surveys, increasing the precision of trend estimates. The performance of sampling designs similar to the one presented in this paper has been evaluated previously (Urquhart et al. 1998; Urquhart and Kincaid 1999). The rotating-panel design used by ODFW had not been evaluated, however, since the monitoring program was initiated in 1998, and the design has not been employed by any other program for as long a period. Our approach is heuristic insofar as the statistical model we employ, though plausible, is a simplified way of thinking about trend in a complex system shaped by innumerable factors that we cannot readily dissect. These analyses can reveal relationships and provide clues to underlying mechanisms dictating change. We note that the monitoring program described here was designed to detect trends at regional scales, but habitat changes often occur at local resolutions and on incremental time scales. The results we note here are scale-dependent. The resolution of the sample of sites is a function of both the sampling intensity and spatial balance of the design. We can only make inferences at the extent for which the sampling design was intended and data were collected.

Implications for Monitoring

Statistical model and design performance.—There has been substantial discussion about the appropriate statistical model to detect trends (Stow et al. 1998; Larsen et al. 2004; Seavy and Reynolds 2007). Our analysis and results do not preclude the existence of alternative spatial structures that may exist in the data. We offered an analysis that was appropriate given the data set and the scale at which habitat may be changing relative to coho salmon population and life history dynamics (Kostow 1995). Following Larsen et al. (2004), our model posits four independent variance components associated with site, year, site-by-year interaction, and residual-error variability. While all four sources of variability influence trend detection capability, each does so differently. Thus, a supplementary objective of this study was to determine the relative contribution of each source. For each of the habitat features examined, site (i.e., spatial location) accounted for the majority of the variability and year accounted for the least. Considering that sites were sampled across many different regions over a wide range of local landscape features and stream types, this is perhaps not surprising. The relative proportion of variation attributable to residual error was of intermediate magnitude in these data sets and differed only slightly among the habitat features analyzed, indicating that the relevant sources of error variability and processes causing observed data to vary (e.g., errors in measurement, sampling, data entry, etc.) were consistent among the habitat features. Because the ODFW sampling plan incorporated within-year site revisits we were able to partition the effect of residual variance and increase the precision of our trend estimate (Larsen et al. 2001).

Coherent temporal variability, expected to result from large-scale or localized fluctuations in multiple environmental factors, was minimal relative to site variability for three of the four habitat features evaluated. In this way, our results were analogous to those found in both Van Strein et al. (1997) and Wagner et al. (2007). Similar to Larsen et al. (2001), however, we found that increased variation can reduce trend detection power. When year variability was minimal, it did not severely influence trend sensitivity and the probability of detecting spatial trends remained high. As we increased year-to-year variation, the probability of detecting trend under each of the two sampling designs assessed was depressed. This indicates that our ability to detect regional trends with either design is sensitive to the magnitude of the year effect. Both Urquhart et al. (1998) and Larsen et al. (2004) commented that while reductions in site, residual, and site-by-year variability can be achieved by adjusting the sampling plan, only two approaches can compensate for a large year effect: (1) investigators can identify and explicitly model important covariates or factors controlling the year effect (e.g., river flow), or (2) if covariates cannot be identified, monitoring duration must be increased for a trend to be detected.

As a further extension to the results found in Urquhart and Kincaid (1999) and Larsen et al. (2001), our results demonstrate that both a conventional design and a panel design can achieve similar power with the passage of time, and trend detection power was generally comparable under the two designs. The potential for the panel design was discussed in Larsen et al. (2004) but had not been demonstrated until our study. We found that power is sensitive to the duration of the monitoring effort. The power to detect a 1% effective trend after 15 years was approximately 93% for the panel design and 95% for the conventional design. Similarly, to detect a 2% effective trend after 10 years, the probability was 98% under either design. The advantage of the panel design is a substantial increase in the number of sample sites across the landscape, increasing the ability to assess status with little or no loss in trend detection power in relation to the alternative design. Though our power simulations are modeled on one set of field data, it appears the panel structure is in this sense a robust design that does not lose much trend detection power despite fewer annual repeat sites. As Urquhart and Kincaid (1999) and Urquhart et al. (1999) observe, the cyclical patterns of site revisits across years can be nearly as powerful as an annual design after three cycles.

While we provide a readily accessible and applicable approach to estimating trend detection power, we were forced to modify our approach, given the complications we encountered related to the size and complexity of the original data set. We managed this complexity through simulations that allowed us to incorporate the sampling variability of the variance component estimates used in our original test. We adjusted the sampling design, the monitoring period or duration, the hypothesized trend, and the variability structure. The simulations were based on an assumed linear model, as described above, which included intercept and variance component parameters based on those estimated from the observed data. Although our hypothesized trends were consistent with the empirical results of this study, it is misleading to compare these two analyses because they addressed different questions. The objectives of the simulations were to estimate the power to detect trends and to examine the sensitivity of trend detection given changes in the sampling design and year-to-year variability. Simulations were not used to learn more about the observed data sets, but instead to compare the trend detection efficacy of different sampling plans under realistic scenarios.

In general, the type I error rate estimates tended to be high, suggesting that the associated power estimates were probably inflated. These estimates were inconsistent, however, so it is difficult to make meaningful comparisons among different monitoring periods, sampling designs, and variability structures. Meaningful comparison depends on a common-baseline type I error rate. Nevertheless, values 0.133 and above are suspect because they are more than three standard deviations away from the nominal α level of 0.10. Generalizations should be limited when comparing scenarios with such error rate estimates.

Trend detection in freshwater habitat and relationships to salmonid populations.—While numerous drivers influence salmonid population trends, the quality of freshwater habitat consistently contributes to overall population growth, particularly in the presence of fluctuating marine climate conditions and intrinsic population dynamics. Buhle et al. (2009) were able to isolate several critical factors influencing coho salmon productivity in 15 coastal populations; they demonstrated that habitat quality as it relates to juvenile salmon carrying capacity had a significant and positive effect on productivity during a time of variable ocean conditions and declining hatchery influence (1990–2003). The results noted here may support the conclusion that, with adequate freshwater habitat, Oregon coast salmonids can remain viable during years of relatively poor ocean conditions (Chilcote 1998; Chilcote et al. 2005; Suring et al. 2008), retain their resiliency, and respond positively during cycles of improved ocean productivity (Moore 2009). Moreover, we expect improvements in freshwater habitat to reflect higher adult base productivity during poor ocean years, as hypothesized by Lawson (1993), emphasizing the importance of not only habitat quality (Coronado and Hilborn 1998) but the increasing availability of previously inaccessible habitats.

Studies conducted at local and regional scales have inferred contributory linkages between fluctuations in particular habitat characteristics and large-scale natural processes and management effects (Czarnomski et al. 2008; Jorgensen et al. 2009). Owing to the complex nature and interdependent relationships at multiple spatial scales among stream habitats, however, a change in a particular habitat is often difficult to match to an individual stressor or event (Allan 2004; Kaufmann and Hughes 2006). For example, studies have described the reduction of pool habitat within the Columbia River basin (McIntosh et al. 2000), simplification of channels due to splash dams and associated activities in coast basins (Miller 2010), and effects of logging on stream habitat and fish in the Alsea watershed (Hall et al. 1987). Overall, the results of this study may suggest that the combination of state and federal protection measures, set at regional scales, may be showing a coastwide effect. Although our study was conducted over a relatively short time frame, the absence of a significant downtrend in habitat condition may signal a change in trajectory compared with historical conditions.

In general, we did not detect statistically relevant trends in the amount of pool area, although we did observe significant differences in pool area among the five monitoring strata (indicated by the intercept parameter of the linear model). Pool area is important from a biological perspective, but this feature had a high site-by-year variance component. We are uncertain whether the high variance resulted from high winter flows in some watersheds, restoration treatments, or the activity of beaver *Castor canadensis*. The high site-by-year variance does not diminish the importance of pools, but more time may be required to detect a significant trend. Certain environmental variables tend to be intrinsically noisier and are sensitive to stresses that prohibit the detection of change (Stow et al. 1998). Investigating additional metrics, and potentially the combined effects of several metrics, may provide more information about the sensitivity of the statistical model relative to the features chosen.

The changes in the amount of fine sediments and wood volume that were observed could be natural and cyclic responses to fluctuating annual weather patterns and hydrologic processes. Owing to the stochastic nature of natural disturbances, which are spatially and temporally dynamic in the Coast Range of Oregon, habitats can shift and change across the landscape (Reeves et al. 1995). Detection of small changes in stream habitats could be representative of these shifting patterns. These alterations in habitat, specifically on the North Coast, were probably affected by a flood that occurred in 1996, before the start of the monitoring program (Taylor 1996), which delivered and redistributed large amounts of wood and fine sediments to streams in this region. This flood was the highest on record for two permanent USGS gauges in the region (Wilson River and Lower Nehalem). Randomized and paired resurveys following the storm revealed that 3% of streams in the North Coast experienced debris torrents and 15% had major channel modifications (K. Moore, ODFW, personal communication). Bank erosion and fine sediments also increased significantly in the North Coast (paired *t*-test of preflood and postflood data; $P < 0.05$), and levels of fine sediments remain substantially higher than in other monitoring areas (Table 4). Since that time, with each winter flow event, fine sediments and instream wood appear to have shifted and have been transported out of the study streams. This region tends to experience the most extreme storms along the Oregon coast.

The lack of downtrends in freshwater habitat condition in coastal watersheds of Oregon is promising. We cannot, however, definitively conclude whether it is due to land-use regulations or a decrease in the effects of timber harvest practices on stream channels, the primary form of anthropogenic disturbance and a major contributory factor resulting in the widespread loss of habitat prior to the 1990s. Aquatic habitat has benefited from the initiation of the Oregon Forest Practices Act (1987) and the Northwest Forest Plan (1994), which significantly modified practices on private lands and reduced timber harvest on public lands; however, the rate and extent of timber harvest on private industrial lands has remained relatively steady over the monitoring period (Oregon Department of Forestry 2010). This could account for the increase in sedimentation that was detected in the Mid-South Coast region, which has the highest proportion of private industrial forests relative to the other monitoring strata (Burnett et al. 2007). Numerous studies have documented adverse effects from forest harvest and road building on stream networks and in-stream sediment delivery processes (Croke and Hairsine 2006; Beschta 1978). Specific increases in fine sediment do not always indicate negative impacts on salmon, however. For example, an increase in beaver pools may trap fine sediments but also increase rearing capacity and survival for juvenile salmonids during the winter (Nickelson 1992). This habitat feature must, therefore, be placed in context to be interpreted appropriately.

The pertinence of these analyses to policy makers, biological review groups, and resource managers is central to the monitoring effort. The scale and context of the results posit an understanding of overall species risk via in-stream habitat condition, informing conservation and recovery efforts. As we report in this study, trend detection rates will vary over different time frames, trend magnitudes, and degrees of annual variation. Acknowledging the caveats to evaluating habitat change can guide the design of other monitoring programs. Subtle changes in stream conditions, as represented by these four features, provide an opportunity to prioritize stream improvements and restoration at local scales. Resource managers and watershed groups that implement holistic stream and terrestrial improvements will more likely effect change and influence stream habitats beyond the treatment area. These changes in habitat will have a higher probability of being detected at broader spatial extents through monitoring and contribute to assessments of watershed health and species persistence.

Conclusions

Considerable resources are required to collect consistent long-term monitoring data across the landscape, which is why trends in freshwater habitat have rarely been quantitatively assessed (Bernhardt et al. 2005; Lovett et al. 2007). Our results lend support to the use of a panel design when challenged with balancing logistic (and financial) constraints against the technical goals of a monitoring program. It is an effective approach when the desired outcome is to optimize status estimates and detect trends. Variance partitioning and power estimation provided confidence to assess the absence or magnitude of trends in physical indicators of salmonid habitat. This approach allows coupling the overall status and trend assessment at the regional scale with more detailed monitoring of change at a local scale. Although the time frame demonstrated here is short relative to large-scale habitat change, regular reporting is valuable in order to reassess existing methodologies and hypotheses about freshwater habitat and its influence on salmonid populations. Although we speculated on potential localized effects that may have contributed to our results, we suggest that it is advantageous to design the monitoring program to assess status and trends at a regional scale, while relying on more directed or controlled small-scale research studies to evaluate specific cause and effect relationships.

Freshwater habitat quality is intrinsically linked to the fate of salmonid populations in coastal watersheds in Oregon, influencing productivity and survival. The trend results indicate subtle changes in habitat over the monitoring period, reflecting a generally stable condition. This effect could imply a reversal of what was considered a long-term downward trend in aquatic habitat condition. Resource management actions that restore habitat complexity and repair ecosystem processes to promote connectivity of high-quality habitats will maintain this course and produce long-term benefits to salmonids.

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