DEVELOPING APPROPRIATE SEDIMENT–RELATED WATERSHED CONDITION INDICATORS FOR NATIONAL ENVIRONMENTAL POLICY ACT ANALYSES AND BIOLOGICAL ASSESSMENTS IN THE SOUTH FORK SALMON RIVER BASIN

July, 2005

Rodger L. Nelson, Fisheries Biologist
David C. Burns, Forest Fisheries Biologist

EXECUTIVE SUMMARY

In 2003, The Payette National Forest, along with the Boise and Sawtooth National Forests, which together comprise the Southwest Idaho Ecogroup, published their second Land and Resource Management Plans (LRMPs) under the Forest and Rangeland Renewable Resources Planning Act and the National Forest Management Act. With respect to fish habitat, the revised LRMP identified various "Watershed Condition Indicators" (WCIs) and specified numeric values considered to represent aquatic systems that were "Functioning Appropriately" (FA), "Functioning at Risk" (FR), and "Functioning at Unacceptable Risk" (FUR). Several of these WCIs have generic values to define the functional categories, which was recognized by the National Marine Fisheries Service (NMFS or NOAA Fisheries) during formal LRMP consultation under the Endangered Species Act. Consequently, the Biological Opinion issued by NOAA Fisheries specified, as legally required terms and conditions for issuance of the Biological Opinion and associated incidental take authorization, continuation of the Payette's sediment monitoring and a closer look at the sediment WCIs and development of more appropriate values for the South Fork Salmon River (SFSR).

We briefly reviewed the major natural and management actions that we know affect sediment yield and, potentially, streambed sediment conditions (wildfire, logging, livestock grazing, mining, road building, and road decommissioning and obliteration) in pristine and developed watersheds. This review was provided to show that natural events like wildfire have a quantitatively different, less predictable effect on increases in streambed sediment. In land management planning, these differences are expected to require differences in analyses of effects to streams and fish.

This report then discusses the sediment monitoring on the Payette National Forest and Boise National Forest, which has included interstitial and surficial analysis in small, wadable streams across the Forest in developed and undeveloped watersheds, as well as monitoring of intragravel sediment in spawning areas traditionally used by anadromous fish in the South Fork Salmon River and Chamberlain basin. Monitoring began shortly after the floods of 1964-65 in the South Fork Salmon River to evaluate recovery as the Boise and Payette National Forests implemented watershed improvement actions, and was expanded to include determination of baseline intragravel and interstitial conditions to determine appropriate standards and guides for project planning. The original Payette National Forest LRMP specified interim standards and anticipated using the results of this monitoring effort to refine them during subsequent LRMP revisions. The revised LRMP did not build on South Fork Salmon River monitoring when promulgating WCIs, leading to the need to revisit them, which this report does.

We have analyzed all of the available interstitial monitoring data from reference (*i.e.*, undeveloped or minimally developed) sites on the east side of the Forest through 2003, which includes cobble embeddedness measurements, free matrix counts, and surface fines estimation, to (1) determine average conditions in essentially pristine watersheds, (2) to determine what conditions, including inherent variability in indicators, we should therefore expect absent development, (3) to compare these conditions among watersheds to investigate inherent differences due to location, and (4) to use this information to determine what sediment conditions would provide reasonable indicators for the three functional groups (FA, FR, and FUR) identified in the revised LRMP. In contrast, intragravel sediment conditions in major spawning areas for anadromous species were pooled, based on lack of clearcut differences in reference and non-reference areas in statistical comparisons, to estimate appropriate intragravel conditions in the SFSR.

Although we relied on nonparametric statistics, we were able to detect statistically significant differences in surficial and interstitial sediment indicators (surface fines, cobble embeddedness, and free matrix counts) between sites in granitic watersheds and those in watersheds with largely volcanic and metamorphic parent materials. With spawning area fines at depth (*i.e.*, in core samples) we also showed that the developed Secesh River sites were likely more similar to sites in the undeveloped Chamberlain Basin with respect to fines smaller than 6.3mm diameter than sites in the upper South Fork Salmon River; however, this comparison was reversed for fines smaller than 0.85mm.

Using our evaluations of reference conditions and variability we determined median conditions and variability of these sediment-related fish habitat indicators under essentially natural conditions in granitic watersheds. Using this information, and with reference to information in the fisheries literature relating levels of fine sediment abundance and salmonid production and survival, we estimated what conditions we could expect in streams functioning at the three categories developed in the revised LRMP (FA, FR, and FUR). While we retained the basic functional group arrangement for indicators as contained in the LRMP, we have proposed four major categorical changes (Table A, next page), including: (1) modifications to the indicator names; (2) combining indicators for salmonids where appropriate and rearranging species associations (*e.g.*, the intragravel WCI was changed to anadromous fish rather than bull trout); (3) suggesting using free matrix counts in preference to cobble embeddedness measurements for interstitial conditions; and (4) eliminating or relegating surface fines to a support role. Our suggested WCI values were based on local data, using the distributions of site averages to modify those in the revised LRMP, and we described measurement methods as well. These proposed WCIs incorporate inherent variability so that risks to the aquatic system can be minimized when Forest projects are planned and implemented in the granitic portions of the SFSR. We need to stress that these criteria should not be used outside the geographic area analyzed unless there are local data that indicate that they are appropriate, and data must be collected using these methods to be compatible with these criteria. While not specifically suggested here, this document does provide a framework for evaluating conditions in the EFSFSR, which is geologically different than the granitic portions of the SFSR (*i.e.*, the Secesh River and upper and lower SFSR).

We recognize that the proposed interstitial criteria appear to make higher sediment levels acceptable than usually supposed; we believe that this is at least partly a consequence of attempting to work with artificial categories that may not adequately model the real world. In other words, we have had to place values of continuous variables in somewhat arbitrary categories. We have tried to accommodate this conundrum by allowing evaluation of conditions at multiple scales, the simplest being a single sampling that has more risk of incorrectly interpreting actual conditions, to multiple sampling in space or time that is more likely to estimate true conditions; the former values are more conservative than the latter.

Table A.—Proposed revised watershed condition indicators (WCIs) for table B-1 of the revised PNF LRMP, expressed as functions of surface or depth fines, cobble embeddedness, and free matrix counts. Use of this table assumes the following: (1) all data QA/QC procedures from sediment reports were used; (2) the CE-FM regression indicates valid data were collected; (3) where multiple metrics are available, the ones indicating the highest sediment levels are used; (4) the longest time interval available is used with the most recent data; (5) data from the nearest downstream sites are used; and (6) analysis not clearly discriminating between two functional classes indicates that the lower class be used.

Pathways and WCIs	***Functioning Appropriately***	***Functioning at Risk***	***Functioning at Unacceptable Risk***
Interstitial Sediment Deposition (all listed fishes in tributary systems)	Adequate interstitial space is indicated by: (a) Any single **measured** mean embeddedness value less than or equal to 24%.	Reduced interstitial space is indicated by: (a) Any single **measured** mean embeddedness value between 24% and 32%.	Inadequate interstitial space is indicated by: (a) Any single **measured** mean embeddedness value over 32%.
	OR (b) Any single mean free matrix count over 27%	**OR** (b) Any single mean free matrix count between 17% and 27%	**OR** (b) Any single mean free matrix count less than 17%
	OR (c) A five-year mean **measured** cobble embeddedness level of 32% or less	**OR** (c) A five-year mean **measured** cobble embeddedness level of 32% to 42%	**OR** (c) A five-year mean **measured** cobble embeddedness level greater than 42%
	OR (d) A five-year mean free matrix count of 17% or more.	**OR** (d) A five-year mean free matrix count of 11% to 17%.	**OR** (d) A five-year mean free matrix count of less than 11%.
Interstitial Sediment Deposition (other fish species: *i.e.*, red band, rainbow, wood river sculpin, *etc.*)	For salmonids, use same as for listed species, develop criteria for other species as needed.		
Intragravel Quality (in areas of spawning and incubation for anadromous fishes)	High intragravel quality is indicated by: (a) 5-year mean fines < 6.3 mm concentrations at depth of 28% or less with no more than two years between 28% and 36%.	Moderate intragravel quality is indicated by: (a) 5-year mean fines < 6.3 mm concentrations at depth 28% to 36% with no more than two years > 36%.	Low intragravel quality is indicated by: (a) 5-year mean fines < 6.3 mm concentrations at depth of 36% or more.
	OR (b) 5-year mean fines < 6.3 mm concentrations at depth between 28% and 36% with a **decreasing** trend over at least 10 years.	**OR** (b) 5-year mean fines < 6.3 mm concentrations at depth between 28% and 36% with an **increasing** trend over at least 10 years.	**OR** (b) 5-year mean fines < 6.3 mm concentrations at depth 36% or more with an **increasing** trend over at least 10 years.
		OR (c) 5-year mean fines < 6.3 mm concentrations at depth of 36% or more with a **decreasing** trend over at least 10 years.	
Substrate Embeddedness (Bull trout rearing areas. Spawning and incubation areas are addressed under the Sediment/Turbidity WCI)	Replaced with Interstitial Sediment Deposition for Listed Fishes WCI above		

CONTENTS

List of Tables

LIST OF FIGURES

INTRODUCTION

The first Land and Resource Management Plan (LRMP) for the Payette National Forest (PNF) pursuant to the requirements of the Forest and Rangeland Renewable Resources Planning Act of 1974 (RPA, 16 USC §§ 1600–1614) as amended by the National Forest Management Act of 1976 (NFMA [and subsequent amendments]), and their implementing regulations was published in 1988. This plan, called a "Land and Resource Management Plan" (LRMP) or, simply, the "Forest Plan," was intended to provide the framework for land management activities during the lifetime of the plan (*i.e.*, the planning horizon) by defining desired conditions for various resource groups, setting general goals consistent with those desired conditions, and establishing planned objectives designed to achieve those goals within an established time frame. Standards and guidelines imposing specific constraints on land management actions taken pursuant to achieving resource goals in order to insure environmental protection were also described in forest plans.

The expected lifetime of the "old" Payette National Forest (PNF) LRMP under then-current regulations was approximately 10 years but not to exceed 15 years (16 USC 1604[f][5][a]). Although LRMPs could be revised on a shorter schedule if changed conditions indicated that revision was appropriate, the full 15 years elapsed prior to publication of the revised LRMP for the PNF. During this period, social and biological, and physical conditions have changed dramatically with, among other things, listing of three fish species under the Endangered Species Act of 1973, as amended (ESA, 16 USC §§ 1531–1544) and wildfires covering a large proportion of the Forest's 2.3 million acres. Revision of our LRMP was clearly needed from both a legal and a practical perspective. Consequently, in 2003, The Payette National Forest, along with the Boise and Sawtooth National Forests, which together comprise the Southwest Idaho Ecogroup, published their second Land and Resource Management Plans under the Forest and Rangeland Renewable Resources Planning Act and the National Forest Management Act. These provide an analytical mechanism for determining salmonid habitat function with respect to sediment-related indicators for anticipated projects that have the potential to alter instream sediment.

The potential effects of increased fine sediments caused by ground disturbing land management activities that alter natural sediment delivery rates to streams are highly diverse. While it is beyond the scope of this document to discuss those potential effects in detail[1], it is important to note that the old LRMP recognized the fact that Forest management activities can alter natural sediment mechanics in managed watersheds with potential deleterious effects in rivers and streams. This was abundantly clear after the dramatic flood events of 1964-65 that inundated important Chinook salmon and steelhead spawning grounds in the South Fork Salmon River (SFSR) with fine sediment proved the need to better understand the interactions of mechanical disturbance, hillslope hydrology, instream sedimentation, and survival of aquatic organisms, particularly salmon and steelhead because of their great cultural, aesthetic, and economic value. Consequently, although understanding of inherent sediment delivery processes and natural levels of streambed fine sediments was not well developed, the old LRMP promulgated standards and guidelines relative to whether increased deposition of fine sediments would be allowed. Concurrently, the old LRMP anticipated refinements to numeric sediment criteria and required both continued monitoring of Chinook and steelhead spawning areas in the SFSR and the implementation of new sampling in other

[1] A review of the various potential effects of fine sediments on salmonids can be found in Chapman and McLeod (1987).

areas to more accurately determine appropriate sediment criteria for management purposes.

The revised LRMP took a functional view of sediment conditions using ranges in index values adapted from the National Marine Fisheries Service (NMFS, more recently "NOAA Fisheries Service"[2]) and U.S. Fish and Wildlife Service (USFWS) matrices identified in the 1995 and 1998 PACFISH/INFISH biological opinions (BOs) (NMFS 1995, 1996[3]; USFWS 1998). Thus, ranges of expected values for sediment indices, called "Watershed Condition Indicators" (WCIs) were defined for three functional categories: "Functioning Appropriately" (FA), "Functioning at Risk" (FR), and "Functioning at Unacceptable Risk" (FUR). Fish habitat potentially affected by proposed projects was to be classified into one of three classes and the expected effects of the project to each indicator were to be analyzed. These categories place substantial constraints on project design because proposed projects are prohibited from moving FA indicators to either FR or FUR and must not retard attainment of FA when pre-implementation indicators are either FR or FUR.

The revised LRMP also anticipates the need to develop site-specific modifications of WCIs for some projects where environmental analysis indicates that they are inappropriate (LRMP IV-2 ¶3), and this has been done for some projects. Because we have an extensive sediment monitoring database from the SFSR, we are in a unique position to derive suitable sediment-related WCIs for that watershed based on inherent, local ranges for the various indicators. NOAA Fisheries Service, in fact, has promulgated a requirement that we do so in the biological opinion (BO) they prepared pursuant to ESA consultations on the revised LRMP (NMFS 2003)[4].

We have performed this analysis specifically to correct WCIs for use in the granitic portions of the SFSR, which excludes the East Fork SFSR (EFSFSR). However, the data used were drawn from the Chamberlain Basin and parts of the Edwardsburg area as well, both of which are also predominantly granitic watersheds. It is important to avoid extrapolating the results of one study outside the range of the data analyzed, but it seems reasonable to suppose that the results of this effort would minimally apply to the Chamberlain Basin and upper Big Creek areas as well as to the SFSR, and may be better in granitic areas generally than the default WCIs in the revised LRMP; however, extrapolations outside the geographic range of these data should be accompanied by local data that support their applicability.

[2] This document will mainly use NMFS, the official name of the organization, when acronyms are appropriate.
[3] The U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service is now more commonly referred to as the NOAA Fisheries Service; we will use both names interchangeably.
[4] The U.S. Fish and Wildlife Service also issued a biological opinion on plan revision (USFWS 2003) but issued no special terms and conditions.

SEDIMENT, AQUATIC ECOSYSTEMS, AND LAND MANAGEMENT

OVERVIEW

All streambeds contain sediment. Indeed, streambed sediment comprises perhaps the most fundamental habitat component of lotic ecosystems. Streambed sediments are the result of ambient erosion processes and may be alluvial or colluvial in nature, with their particle size distributions being primarily a function of underlying geology and lithology. In undisturbed landscapes, streams have adjusted to the prevailing characteristics of the geologic cycle and disturbance regime, with sediment inputs approximately balancing outputs such that overall conditions appear to be relatively stable. Streams are not stable, of course, but obvious changes, barring an extreme event like a landslide, occur at a rate that is essentially undetectable to the casual observer. These continual yet subtle changes maintain the variability and diversity in habitat conditions that we see as natural conditions. Aquatic ecosystems, in turn, have adapted to these natural conditions, including their inherent, sometimes capricious, variability. Aquatic organisms have adapted to their environment through a variety of mechanisms too numerous to mention here, but suffice it to say that some adaptations may be very site-specific. Often, local adaptations, like subtle changes in aquatic conditions, are not particularly obvious and may require specialized genetic methods to demonstrate; nonetheless, they may be very important to the persistence of local populations. A good general review of the role of sediment in streams has been prepared by Waters (1995).

While the above is generally true, it is equally obvious that the natural range of conditions may periodically deviate from conditions to which organisms are adapted. In the short term, such disturbances, if they are short-lived, may improve overall conditions or may reverse deteriorating conditions; if conditions stay outside tolerable ranges, they may drive local organisms to extinction. Climate change is an example of a potential long-term process that is probably changing species distributions, though stressors like this often act over such lengthy temporal scales that we fail to understand or even notice them. Human activity, however, can introduce disturbances that exceed the tolerance of populations of organisms and can occur with sufficient magnitude or over sufficiently large geographic scales to drive organisms to extinction; thus, many species, including several on the Forest, have been listed under the ESA. In our case, logging, livestock grazing, mining, road building, and road decommissioning and obliteration, for example, can act individually or collectively to modify aquatic conditions such that they fail to provide adequate habitat for native aquatic organisms. We have some experiences with various natural and management disturbances, which are discussed below.

WILDFIRE

Wildfire is common on the Payette and Boise National Forests, and since 1988 nearly a million acres have burned on the PNF alone. While most fires are successfully controlled during initial attack, large (over 100 ac) fires occur regularly and can easily affect larger land areas over a shorter time than any anthropogenic activity. A treatise on the potential effects of wildfire on salmonid habitat is beyond the scope of this report, but a good general summary can be found in Swanston (1991). The principal effect of wildfire is to remove vegetation that serves to stabilize soils and can increase water yield and erosion in burned watersheds; if riparian vegetation is killed, its ability to buffer streams from increased erosion may also be diminished.

Despite the fact that wildfires appear to be so destructive, deleterious effects of wildfire on salmonid fishes and their habitats have rarely been documented, particularly over the long term. For example, Rieman *et al*. (1997) studied the effects of large wildfires on the Boise National Forest (BNF) in 1992 and 1994 where both direct fish mortality and extensive habitat changes were documented, and discovered that numbers of both redband trout (*Oncorhynchus mykiss*) and bull trout (*Salvelinus confluentus*) numbers returned to pre-fire levels within three years. Although we have pursued no formal studies of the recovery of fish following the large wildfires on this Forest, we have some indirect evidence that the overall effects of wildfire are likely beneficial and probably different than the effects of anthropogenic vegetation removal actions (*i.e.*, logging, which is often seen as an anthropogenic action with effects similar to fire). Potential beneficial effects of fire include coarse sediment input, large wood recruitment, nutrient additions, and simply availability of more water. We have performed aquatic ecology monitoring of fire effects on the Forest and have documented time trends of fish habitat indicators on the Forest that generally show that wildfire has no measurable long-term effect on aquatic ecology (Bowman *et al*. 1998; Bowman and Minshall 1999; Minshall *et al*. 1994; Royer *et al*. 1995, 1997; Royer and Minshall 1996).

Figure 1.—Trail Creek, North Fork Payette River watershed in October of 1994, shortly after it burned in the Blackwell–Corral Complex fire.

We have visual evidence that there may be beneficial effects of fire from a photomonitoring site on Trail Creek. Trail Creek is a tributary of the North Fork Payette River, and it burned intensely in the Blackwell–Corral Complex fire1994 (Zuniga *et al*. 1994). Conditions immediately post-fire (Figure 1) included blackened hillsides and streambanks, some deadfall in the creek, and a nearly completely dead forest canopy. After 8 years[5], however, it is clear that riparian vegetation has recovered (Figure 2), but what is really interesting is that the streambanks and bed appear to be much coarser now than they were in 1994. We hypothesize that increased water yield following the fire produced flows sufficient to cleanse the streambed that were also not large enough to dramatically reorganize the channel despite enduring some years where flood flows were common (*e.g.*, 1997).

Figure 2.—Trail Creek, North Fork Payette River watershed in August 2002, 8 years after it burned in the Blackwell–Corral Complex fire

[5] The entire time series, which does not include all years following the fire but does show the beginnings of recovery, can be seen on the PNF Fisheries Program FSWeb site (http://fsweb.payette.r4.fs.fed.us/units/fish.web/FishStart.htm) by going to the "Reports" page and looking under "Time Series Photography" on the main form menu.

The indirect evidence we have that wildfire effects on streambed sediments are probably different than anthropogenic activities, at least on this Forest, comes from an analysis of equivalent clearcut area (ECA) and fish habitat conditions that we performed recently (Nelson *et al*. 2004a). This analysis showed that ECA (which is essentially an index of canopy loss that could occur from logging or fire) was positively correlated with streambed fine sediments in roaded and logged watersheds, but not in less-roaded burned watersheds. This suggests that although wildfire may lead to increased sediment yield as a consequence of increased water yield, it does not translate directly into increased sediment deposition in streams.

TIMBER HARVEST AND ROAD CONSTRUCTION

We have to discuss the potential effects of timber harvest, with which we include the road construction that usually accompanies it, because we have seen examples of them on this Forest. Potential effects are reviewed in Chamberlin *et al*. (1991) and Furniss *et al*. (1991) and are not reiterated here; however, we point out that we believe that the road construction component is probably the principal factor causing timber harvest to have fundamentally different effects on aquatic systems than wildfire. While timber harvest removes hillslope stabilizing vegetation, road construction removes the productive A and B horizons of forest soils and replaces them with the unproductive coarse-grained mineral horizon and disrupts the normal hydrologic function of the watersheds affected in various ways. There has been considerable study in the Idaho Batholith showing both increased sediment yield and deposition as a result of logging and associated road construction, though the linkage between yield and deposition are poorly developed (reviews of these processes and interactions in the context of our area can be found in Cline *et al*. 1981 and Stowell *et al*. 1983).

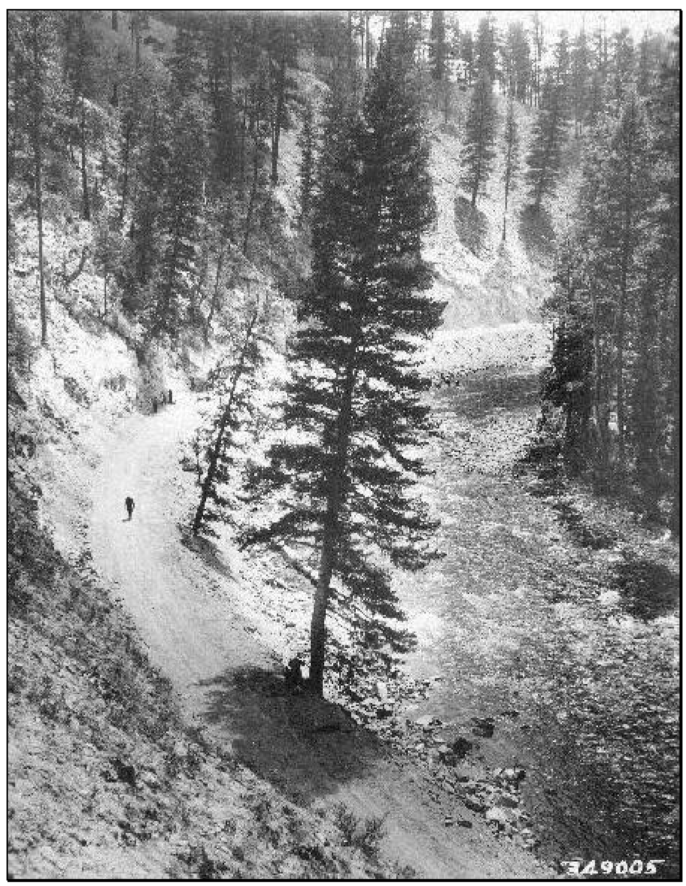

Figure 3.—Fresh roadcut of the South Fork Salmon River road near Buckhorn Bar in 1937.

In the South Fork Salmon River, the principal focus of this report, road construction had begun by at least 1937 (Figure 3). As is clear from this image, the topography tends to be quite steep, and we know that flooding with attendant hillslope failures with transport of the weakly consolidated granitic soils and larger materials to the river and its tributaries was a relatively common phenomenon. Logging and logging road construction began in earnest sometime in the early 1950s (Figure 4, next page) and continued until 1965, accompanied by increasing sediment yields over natural levels. In the winter of 1964-65, heavy warm rains fell on a well-developed snowpack and caused serious hillslope failures throughout the upper SFSR and lower Secesh River watersheds; these problems were exacerbated in the spring by additional heavy rains on saturated hillside soils (Jensen and Cole 1965). This situation has been discussed in the literature, with the most recent discussion of salmonid habitat condition and trend being Platts *et al*. (1989).

When the "Christmas Storms" of 1964-65 hit the SFSR watershed, there were some 600 miles of roads in the watershed (Platts and Megahan 1975), often at very high density (Figure 5). The floods were, of course, natural events, but the destabilization of the hillsides caused by the extensive road network and the acres of exposed mineral soil on road surfaces and cut and fill slopes undoubtedly increased the amount of material that was ultimately delivered to the river and which ultimately inundated the streambed, including several important spawning areas for anadromous salmonids, with duning sand (Figure 6). Afterwards, a logging moratorium was imposed and maintained until the early 1980s when limited entries were allowed, and a wide range of watershed rehabilitation projects and streambed sediment monitoring programs were started. Now, about 500 miles of road have been closed; most of these have grown over but many miles have been obliterated. Monitoring has shown that conditions have and are improving. Platts *et al*. (1989) concluded that the SFSR had successfully removed most of the excess fine sediment deposited by the floods, but had reached an approximately stable equilibrium, and that additional restorative actions in the watershed were required to return the river to conditions approximating those that preceded the floods. We cannot say with certainty what those pre-flood conditions were because there is no objective documentation of them.

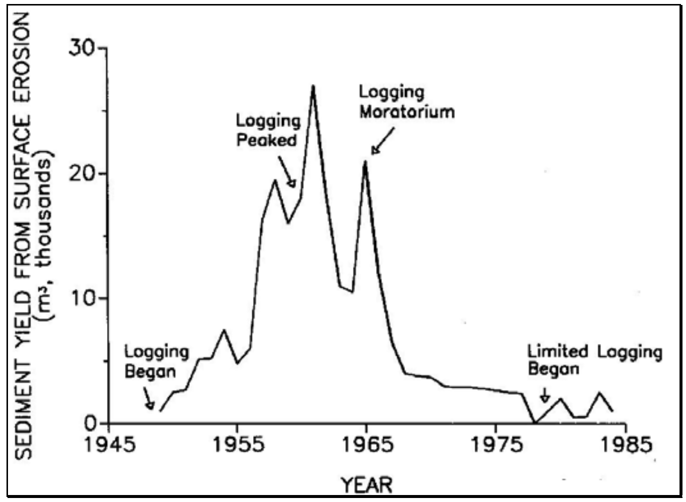

Figure 4.—Increases in sediment yield over time in the South Fork Salmon River watershed (from Platts *et al*. 1989).

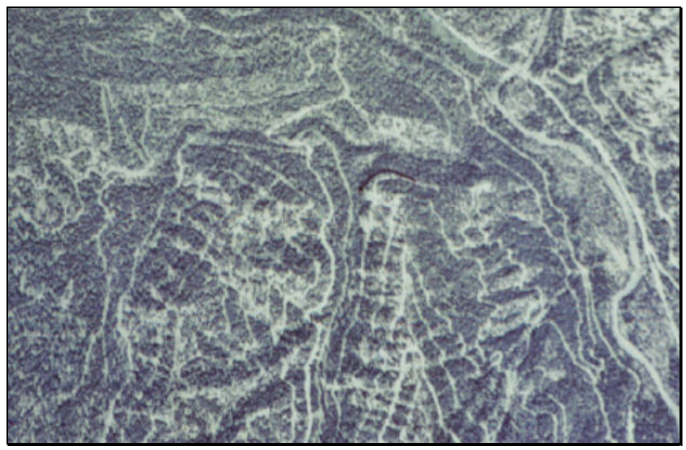

Figure 5.—Aerial view of the road network in Cow Creek, a Secesh River tributary in the early 1960s.

Figure 6.—South Fork Salmon River a short distance upstream of the Darling spawning area in the summer of 1965.

Our monitoring (Nelson *et al.* 1997 *et seq.*), which is done in conjunction with the BNF has shown, however, that conditions are continuing to improve gradually, particularly with respect to very small fine sediments (those smaller than 0.85 mm in particle diameter), and that the appearance of the river has dramatically improved (Figure 7)[6].

Figure 7.—South Fork Salmon River a short distance upstream of the Darling spawning area in the summer of 2004.

MINING

Mining has the potential to disrupt fish habitat, both through sedimentation and direct disturbance of the streambed; a general review of the potential effects of mining on salmonid habitat can be found in Nelson *et al*. (1991). Although very little mining that is likely to increase sediment in streams of the PNF is occurring now, there have been historic situations where we have had problems. The most obvious situation in the SFSR that was known to contribute sediment to the system (Figure 8) was the Stibnite Mining District near the headwaters of the East Fork South Fork Salmon River (EFSFSR). Mines are no longer operating and are the target of extensive watershed rehabilitation efforts. Several sites have been established to monitor sediment conditions downstream of the mines, and latest analyses suggest that streambed surface conditions (*e.g.*, cobble embeddedness levels) are generally good but probably somewhat poorer than would be expected of undisturbed streams in similar geology (Nelson *et al*. 2004d). Other monitoring in the Secesh River watershed, however, has shown that the lingering effects of mine-spoil derived sedimentation can remain for a very long time. Although intragravel sediment cores in Secesh River spawning areas that we routinely sample show relatively low levels of fine sediments, one site that is downstream of an area that was dredged near the end of the 19th century and early in the 20th century (Figure 9, next page) continues to show very poor conditions for survival of salmon and steelhead larvae (Nelson *et al*. 1997 *et seq*.).

Figure 8.—Turbidity in the East Fork SFSR generated by activity at Stibnite Mine at the mouth of the SFSR in the 1980s.

[6] The BNF has maintained a photomonitoring program since 1975 at several established locations; results can be viewed on the PNF Fisheries Program FSWeb site (http://fsweb.payette.r4.fs.fed.us/units/fish.web/FishStart.htm) by going to the "Reports" page and looking under "Time Series Photography" on the main form menu.

LIVESTOCK GRAZING

Grazing can lead to decreased fish habitat quality through trampling of streambanks, decreased bank stability and erosion from exposed streambank soils, and shifts in riparian plant composition with simplification of the streamside plant community. Research, in particular Forest Service research, into the effects of grazing on fish habitat and the efficacy of various grazing management strategies has been extensive; a good general review can be found in Platts (1991).

Figure 9.—Partially stabilized mined area in the Threemile Placer area near Burgdorf, 1999.

Permitted grazing has been discontinued in the SFSR watershed except for some sheep grazing in the headwaters of the Secesh River watershed on the PNF and incidental grazing by horses throughout the watershed. Although sheep grazing is generally considered to be less damaging to salmonid habitat than cattle grazing, we have had instances on the PNF where sheep grazing led to allotment closures, including Little French Creek where historic effects of improper grazing can still be seen (Figure 10). Several sediment monitoring sites have been established on the PNF east side as grazing program mitigation pursuant to consultations with NMFS that initially determined grazing to be likely to adversely affect Chinook salmon. The intent of the monitoring is to determine the ability of our range management to improve or maintain fish habitat conditions in several allotments that contain anadromous fish. While results to date have been largely inconclusive (Zurstadt 2004), they do show that sediment conditions are not deteriorating as we continue to permit sheep grazing.

Figure 10.—Little French Creek in 2001 showing exposed streambank soils and false banks in the process of rebuilding normal structure.

MANAGEMENT IN THE SFSR

The SFSR has hosted all of the forest management activities described above. Although we usually associate the SFSR with a few years of logging in the 1950s and early-1960s, development of the watershed began in the mid- to late-1800s. Around the turn of the century, there were several established mines in the Secesh River and the headwaters of the East Fork South Fork Salmon River (EFSFSR) that stimulated settlement. Increasing population led to the need for food, so livestock were not far behind the early miners. As far back as 1948 we have records of people familiar with the area describing heavy sediment loads in the SFSR that were attributed to overgrazing since about 1916 (Varner

1948). Whether this was due to overgrazing cannot be stated with certainty, but we do know that sand deposits did exist at the mouths of Camp and Phoebe Creeks as early as 1955 (Figure 11). This was shortly after the first commercial logging began in the watershed, and people were beginning to question the wisdom of developing the watershed without extreme caution (Heikennen, no date, on file at PNF Supervisor's Office). We have learned enough about the watershed now to think that some of this sand may well have been natural, resulting from the regular floods to which it is subject (several of these, from 1948 to 1965 are briefly described by Hockaday 1968, records of several others from 1965 through 1997 are on file at the PNF Supervisor's Office) and patterns of bedload movement. However, a notable storm hit after about 15 years of road construction and timber harvest, inundating extensive areas of river bottom with sand (Figure 4). This flooding led to clean-up actions, road closures, a harvest moratorium and monitoring; the streambed sampling has documented the response of the river to these actions (Figure 12)[7]. It is important to note that flooding that was probably similar in many respects to the infamous "Christmas Floods" of 1964-1965 occurred during the winter of 1996-97 (Nelson *et al.* 1998) with a far different response in riverbed sediments.

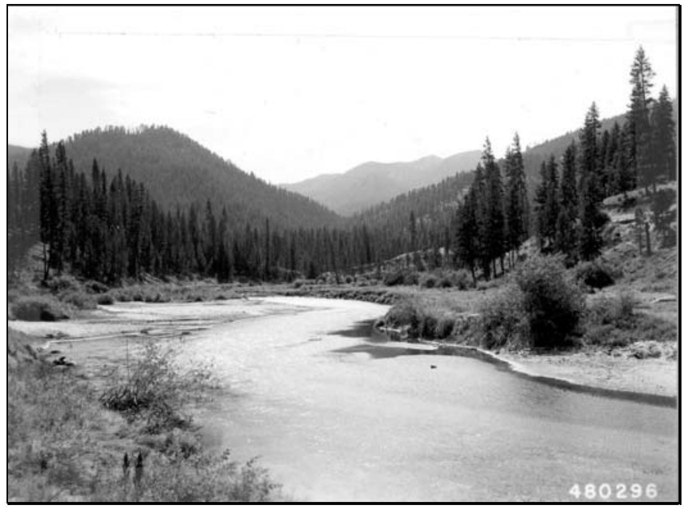

Figure 11.—The SFSR in 1955 just downstream of the Oxbow showing sand deposits at the mouths of Camp and Phoebe Creeks (left).

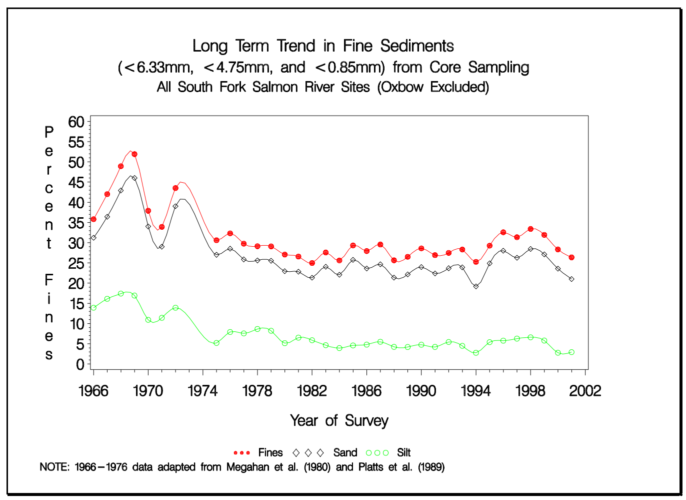

Figure 12.—Proportions of large and small fines from sediment cores in SFSR spawning areas (Oxbow excluded), 1966-2001.

It would seem, because of the steep topography, unconsolidated soils, and variable climate, the SFSR watershed, particularly along its main course, would seem to be unusually sensitive to anthropogenic disturbances. Considerable progress has been made recovering from past management mistakes and nature's whimsy, and we will probably never know what conditions really were before the 1964-65 floods; however, this natural instability clearly constrains additional management actions if the river's unique character is to be maintained. This document is a step in that direction, because it supports adjustment of at least one indicator in the revised LRMP (depth fines) that is set at a level near what the river had in 1970 and would almost certainly be intolerable

[7] Data before 1975 in this illustration were taken from Patts *et al*. (1989) and it is possible, though unlikely, that they are not from exactly the same locations as our monitoring data.

to incubating salmonids. Other indicators will be examined and adjusted as needed to support responsible development while preserving conditions conducive to maintaining the river's natural productivity. We also recognize that the watershed may be inherently functioning at some risk and, consequently, sub-optimal with respect to salmonid production; salmonid production is important, but it does not define this unique ecosystem.

SEDIMENT MONITORING ON THE PAYETTE NATIONAL FOREST

GENERAL MONITORING

Intensive sediment monitoring began in the SFSR following the flooding that occurred in the winter of 1964-65, but an established program with a well-defined protocol for monitoring sediment levels in the river did not begin until 1974. This program was established by the Boise National Forest (BNF), but the SFSR crosses the BNF and the PNF, and this monitoring has recently been pursued as a cooperative effort between the two Forests in coordination with the Forest Service research station in Boise, Idaho. However, monitoring with core sampling at four of the six SFSR sites routinely monitored were specified in the old PNF LRMP (page V-14). Because little objective information about sediment conditions in spawning gravels existed prior to the 1964-65 events, the PNF has established a program of monitoring intragravel conditions in two spawning areas in Chamberlain Basin in the Frank Church River Of No Return Wilderness (FCRONRW) to establish an understanding of inherent spawning area conditions in granitic watersheds (page V-14). An additional four spawning areas in the Secesh River watershed, a developed watershed tributary to the SFSR that was less severely affected by the 1964-65 events, were also specified in the old LRMP (page V-14)[8].

Core sampling, the technique used in the monitoring described above, is an established standard for evaluating sediment conditions and trends in streambed sediments and provides the most complete assessment of substrate composition (Chapman and McLeod 1987). In practice, however, the McNeil method that we use (McNeil 1964) is highly labor-intensive and not well suited to many of the smaller streams on the Forest. Measurements of cobble embeddedness (Kelley and Dettman 1980; Burns 1984; Burns and Edwards 1985), an index that indicates interstitial space in streambed cobbles available to small fish and macroinvertebrates appeared to be the solution to this problem because it was easier to perform, particularly in remote settings, and was suitable to smaller streams and tributaries. Embeddedness monitoring was already being conducted in many streams across the Forest, a program that was formalized in the old LRMP which specified developing an embeddedness monitoring program at 27 locations across the Forest to determine management-induced and potentially inherent embeddedness conditions in a variety of settings (pages V-14 to V-15).

Since 1983, a large number of sites has been established for monitoring sediment conditions, and 18 reports have been produced documenting conditions and trends at these sites[9]. The earliest reports concentrated on the core sampling and embeddedness monitoring, but a new technique, which we call "30-hoop free matrix" (or, simply, "free matrix"), was added in 1988, and a surface fines measurement was added to the free matrix sampling in 1991. It was determined early on that the cobble embeddedness technique was not well suited to streams in watersheds dominated by volcanic geologies (represented on the PNF primarily by basalt) because the very small size of the fine sediment particles allows them to be readily flushed from the system as suspended material rather than being deposited as bedload (Burns and Edwards 1985). While this might suggest that fine sediment is not a significant issue for streams with basalt lithologies, recent evidence suggests that management does increase fine sediments in

[8] Four were specified, fiver were actually established in the Secesh River watershed.

[9] The precise number of sites is not known because of problems with identification of exact locations of some of the older sites and possible correspondence among sites with different identification codes at various times. The most thorough attempt to catalog all sediment sites in granitic watersheds of the Forest is included in Nelson et al. (1997).

these areas as well (Nelson *et al.* 2003b). Consequently, efforts have been undertaken to determine appropriate indices relating sediment to salmonid habitat on the west side of the Forest where volcanic geologies predominate (Nelson *et al.* 2004b).

Updated analyses of sediment conditions and statistically detectable time trends (if any) are documented and discussed in reports that are produced approximately annually for both intragravel and interstitial sampling. Whereas hardcopy reports were the rule prior to 2000, they are now produced primarily as electronic documents with hardcopy available on special request; all sediment reports have been converted to electronic format and posted on the Fisheries Program pages of the PNF FSWEB site. One report, Nelson *et al.* (1997), presented an updated but preliminary estimate of appropriate sediment indices for forest planning, and this report continues that effort with additional years of survey data.

SPECIALIZED MONITORING

The PNF contains areas of concentrated historic and recent mining activity, particularly in the headwaters of the EFSFSR and Middle Fork Salmon River (MFSR) watersheds. Several sediment monitoring sites were established to evaluate conditions in the Edwardsburg (upper Big Creek) area and to monitor continuing effects of the Stibnite (EFSFSR) and Thunder Mountain (Monumental Creek) mining areas. Mining activities have been suspended at Stibnite and restoration activities are underway; some sampling continues to monitor watershed recovery. The situation on Thunder Mountain is unclear, with some recent proposals to restart mining and a contrasting proposal for the PNF to acquire the area; monitoring continues to track changes in conditions resulting from either scenario. The last report attempting to document conditions at all mining-related monitoring sites was Nelson *et al*. (1996); recent reports have discussed only the sites listed in Table 2 below.

Listing of Chinook salmon pursuant to ESA in 1993 led to consultation with the NMFS on ongoing Forest management activities. Initial consultation on the Forest's sheep grazing program, a monitoring program that includes sediment monitoring at several sites in watersheds of primarily granitic geology, most of which were in areas not covered by other monitoring[10]. Typically, both cobble embeddedness measurements and free matrix counts are taken at these sites. Since the beginning of this monitoring effort, there have been several changes (mostly additions recommended by the NMFS) to the suite of sites studied, but there are now 23 sites in the annual program (Table 1). The changes that have occurred can be seen in the reports that are produced each year pursuant to the BOs that have been issued following consultations with the NMFS and, more recently, the USFWS, on the grazing program. The most recent range monitoring report is Zurstadt (2004), which shows that sheep grazing with the applied mitigation measures is not increasing degradation of fish habitat.

Reconstruction of the SFSR Road in 1994 was preceded by consultation with the NMFS regarding potential effects to Chinook salmon. The biological opinion that resulted from this consultation requires us to monitor sediment conditions in the SFSR and in selected tributaries that may be affected by the project. The most recent report discussing the differences between sites affected by the project and their controls (Nelson *et al.* 2003b) indicates that the project has not led to increased fine sediment in potentially affected stream reaches; this conclusion is substantiated by the most recent report discussing core sampling in the SFSR (Nelson *et al.* 2001).

[10] Many sediment monitoring sites to serve the needs of more than one monitoring objective.

PNF ANNUAL SEDIMENT MONITORING PROGRAM

In 1999, a suite of sites was selected to represent the minimum set of sites that should be monitored annually (Table 1, next page). These include sites to be consistent with the old LRMP on the east side of the PNF, which is primarily granitic, as well as sites established to satisfy consultation requirements and monitoring of specific management programs. Some additional sites in the SFSR are also monitored every year, primarily to ensure adequate data for double sampling (Table 2, page 15), but three were established to monitor effects of the SFSR Road Reconstruction Project; the latter have not typically been evaluated in comparisons of road sites because of limited documentation of study design, but will be of value in determining appropriate values for sediment WCIs in the SFSR watershed.

For a period of time prior to 1999, most interstitial monitoring sites had both embeddedness and free matrix sampling, and double sampling showed that embeddedness could actually be predicted from free matrix counts despite differences in sampling design (embeddedness is measured in a much more restricted stratum of instream locations than free matrix, which evaluates conditions over a short reach of stream irrespective of habitat type). Consequently, cobble embeddedness measurement has been restricted to a subset of the interstitial sites in order to avoid duplicated effort while still providing for double sampling, which allows us to estimate embeddedness from free matrix counts and to monitor data quality. The last report that attempted to update information for all extant sediment monitoring sites was Nelson *et al.* (1997); Nelson *et al.* (2003b) contains evaluation of all sites except E050 that are included in the current sampling suite. Most of the sampled streams are in watersheds underlain by granitic rocks; however, the EFSFSR sites have a substantial metamorphic and volcanic parent material component.

There are other sites that have been part of the annual monitoring program but have been discontinued because of inconsistent or insufficient funding (Table 3, page 16). Some of these are "reference" sites that will be used in the analyses discussed below.

The sites where core sampling has been used remain the original set in the SFSR (Corley 1976), Secesh River (Lund 1982), and Chamberlain Creek (Ries *et al.* 1991), and these sites provide an almost continuous record in these areas of 26, 22, and 14 years, respectively[11].

[11] Whereas the core sampling on the SFSR has been uninterrupted, core sampling in the Secesh River and Chamberlain Basin have some missing data.

Table 1.—Principal sediment monitoring sites on east side of the Payette National Forest with annual sampling.

Site Code	Site Name	Stream	Drainage[a]	Status[b]	Data[c]	Report[d]	Purpose[e]	Record[f]
B081	Stolle Meadows	SFSR	USFSR	Developed	SC	Yes	LRMP	1977-2003
B082	Dollar Creek	SFSR	USFSR	Developed	SC	Yes	LRMP	1977-2003
E083	Oxbow	SFSR	USFSR	Developed	SC	Yes	LRMP	1977-2003
E084	Poverty Flat	SFSR	USFSR	Developed	SC	Yes	LRMP	1977-2003
E085	Glory Hole	SFSR	USFSR	Developed	SC	Yes	LRMP	1977-2003
B152	Ice Hole	Johnson Creek	EFSFSR	Pseudo-Reference	SC	Yes	LRMP	1977-2003
E034	Corduroy Jct	Lake Creek	SR	Developed	SC,CE,FM	Yes	LRMP,RG	1981-2003
E048	Burgdorf	Lake Creek	SR	Developed	SC	Yes	LRMP	1981-2003
E033	Threemile	Lake Creek	SR	Developed	SC	Yes	LRMP	1981-2003
E096	Secesh Mdws	Secesh River	SR	Developed	SC	Yes	LRMP	1981-2003
E046	Chinook	Secesh River	SR	Developed	SC	Yes	LRMP	1981-2003
E032	Upper	Chamberlain Creek	CHB	Reference	SC	Yes	LRMP	1981, 1989-2003
E136	Lower	WF Chamberlain Creek	CHB	Reference	SC	Yes	LRMP	1991-2003
E006	Lower	Blackmare Creek	USFSR	Reference	CE,FM	Yes	LRMP	1983-2003
E068	Roadside	Fourmile Creek	USFSR	Reference	FM	Yes	LRMP,RD	1989-2003
E016	Lower	Buckhorn Creek	USFSR	Developed	CE,FM	Yes	LRMP	1983-2003
E023	Original	Fitsum Creek	USFSR	Developed	FM	Yes	LRMP	1988-2003
E054	Lower	Porphyry Creek	LSFSR	Reference	CE,FM	Yes	LRMP	1983-2003
E030	Lower	Elk Creek	LSFSR	Developed	FM	Yes	LRMP	1989-2003
E056	Lower	Pony Creek	LSFSR	Developed	FM	Yes	LRMP	1989-2003
E067	Campground	Fourmile Creek	USFSR	Developed	FM	Yes	RD	1989-2003
E129	Upper	Camp Creek	USFSR	Pseudo-Reference	FM	Yes	RD	1990-2003
E130	Lower	Camp Creek	USFSR	Developed	FM	Yes	RD	1990-2003
B125	Lower	Cabin Creek	USFSR	Pseudo-Reference	FM	Yes	RD	1990-2003
B126	Middle	Cabin Creek	USFSR	Developed	FM	Yes	RD	1990-2003
E057	Lower	Lick Creek	SR	Reference	CE,FM	Yes	LRMP,RG	1983-2003
E062	Lower	Grouse Creek	SR	Developed	CE,FM	Yes	LRMP,RG	1988-2003
E076	Bridge	Tamarack Creek	EFSFSR	Reference	CE,FM	Yes	MN	1983-2003
E050	Mouth	Profile Creek	EFSFSR	Developed	FM	Yes	MN	1988-2003[g]
E132	Upper Sugar	EFSFSR	EFSFSR	Pseudo-Reference	FM	Yes	MN	1991-2003
E133	Lower Sugar	EFSFSR	EFSFSR	Developed	FM	Yes	MN	1991-2003
E086	Mule	Monumental Creek	MFSR	Developed	CE,FM	Yes	MN	1984-2003
E087	Roosevelt	Monumental Creek	MFSR	Developed	CE,FM	Yes	MN	1984-2003
E088	Reference	Monumental Creek	MFSR	Reference	CE,FM	Yes	MN	1984-2003
E035	Nethker Creek	Lake Creek	SR	Developed	CE,FM	Yes	RG	1993-2003
E116	Lower	Nethker Creek	SR	Developed	CE,FM	Yes	RG	1993-2003
E117	Upper	Nethker Creek	SR	Developed	CE,FM	Yes	RG	1996-2003
E071	FH 21	Ruby Creek	SR	Developed	CE,FM	Yes	RG	1993-2003
E142	Upper	Threemile Creek	SR	Developed	CE,FM	Yes	RG	1991-2003
E081	Road 246	Willow Creek	SR	Developed	CE,FM	Yes	RG	1993-2003
W053	Upper	Elkhorn Creek	MSSR	Developed	CE,FM	Yes	RG	1991-2003
W067	Lower	Fall Creek	MSSR	Developed	CE,FM	Yes	RG	1996-2003
W033	Klip Creek	French Creek	MSSR	Developed	CE,FM	Yes	RG	1985-2003
W046	Boundary	French Creek	MSSR	Developed	CE,FM	Yes	RG	1989-2003
W043	Mouth	Little French Creek	MSSR	Developed	CE,FM	Yes	RG	1989-2003
W064	Nameless	Boulder Creek	LSR	Developed	CE,FM	Yes	RG	1993-2003
W061	Boundary	Goose Creek	LSR	Developed	CE,FM	Yes	RG	1993-2003
W062	Meadow	Hard Creek	LSR	Developed	CE,FM	Yes	RG	1993-2003
W063	Boundary	Hazard Creek	LSR	Developed	CE,FM	Yes	RG	1993-2003
W060	Boundary	Mud Creek	LSR	Developed	CE,FM	Yes	RG	1984, 1993-2003
W030	Boundary	Rapid River	LSR	Developed	CE,FM	Yes	RG	1993-2003
W066	Nameless	Thorn Creek	LSR	Developed	CE,FM	Yes	RG	1995-2003
W065	Hells Canyon	Deep Creek	SNR	Developed	CE,FM	Yes	RG	1993-2003
W068	Mouth	Deep Creek	SNR	Developed	CE,FM	Yes	RG	1996-2003

[a] Drainage codes: USFSR – South Fork Salmon River; EFSFSR – East Fork SFSR; LSFSR – Lower SFSR; SR – Secesh River; CHB – Chamberlain Creek; MFSR – Middle Fork Salmon River; MSSR – Mainstem Salmon River; LSR – Little Salmon River; SNR – Snake River.
[b] "Pseudo-Reference" implies that the site is used as a control in a specific comparison, but the site is not necessarily undeveloped; site E057 on Lick Creek is somewhat developed, but still serves as a reference for the Secesh River watershed.
[c] This reflects the data being collected at these sites at the present time; most sites have additional types of monitoring data as well (*e.g.*, wherever FM data are collected there is likely at least some CE data as well). Codes: SC – Sediment Cores; CE – Cobble Embeddedness; FM – 30-Hoop Free Matrix.
[d] Indicates whether data have been included in most recent core or interstitial reports.
[e] Purpose codes: LRMP – Forest Plan (old LRMP); RD – SFSR Road; MN – Minerals; RG – Range.
[f] Indicates period of record for the data type shown or for the data type with the longest recent record; there are some gaps in most records.
[g] Previously discontinued but sampled in 2004.

Table 2.—Other sediment monitoring sites on the Payette National Forest with annual sampling.

Site Code	Site Name	Stream	Drainage[a]	Status	Data[b]	Report[c]	Purpose[d]	Record[e]
E002	Upper	SF Blackmare Creek	USFSR	Reference	FM	Yes	LRMP	1989-2003[f]
E005	Middle	Blackmare Creek	USFSR	Reference	FM	Yes	LRMP	1989-2003
E007	Upper	WF Buckhorn Creek	USFSR	Reference	FM	Yes	LRMP	1989-2003
E008	Lower	NF Buckhorn Creek	USFSR	Developed	FM	Yes	LRMP	1989-2003
E014	Trailhead	WF Buckhorn Creek	USFSR	Reference	FM	Yes	LRMP	1989-2003
E015	Upper	Buckhorn Creek	USFSR	Developed	FM	Yes	LRMP	1989-2003
E017	Upper Crossing	Little Buckhorn Creek	USFSR	Developed	FM	Yes	LRMP	1989-2003
E021	Middle	NF Fitsum Creek	USFSR	Developed	FM	Yes	LRMP	1989-2003
E022	Lower	NF Fitsum Creek	USFSR	Developed	FM	Yes	LRMP	1989-2003
E024	Lower	Fitsum Creek	USFSR	Developed	FM	Yes	LRMP	1989-2003
E098	Upper	NF Buckhorn Creek	USFSR	Developed	FM	Yes	LRMP	1989-2003[g]
E099	Canyon	Fitsum Creek	USFSR	Developed	FM	Yes	LRMP	1990-2003
E124	Middle	Fitsum Creek	USFSR	Developed	FM	Yes	LRMP	1990-2003
E138	Upper	NF Fitsum Creek	USFSR	Developed	FM	Yes	LRMP	1990-2003
E028	Middle Fork	Elk Creek	LSFSR	Developed	FM	Yes	LRMP	1989-2003
E029	Mouth	WF Elk Creek	LSFSR	Developed	FM	Yes	LRMP	1989-2003
E031	Yellow Jacket	Elk Creek	LSFSR	Developed	FM	Yes	LRMP	1989-2003
E039	Lower	Sheep Creek	LSFSR	Reference	FM	Yes	LRMP	1989-2003
E055	Upper	Pony Creek	LSFSR	Developed	FM	Yes	LRMP	1989-2003
E143	Lower Middle	Elk Creek	LSFSR	Developed	FM	Yes	LRMP	1990-2003
B127	Upper	Cabin Creek	USFSR	Reference	FM	Yes	RD	1990-2003
E128	Lower	Fourmile Creek	USFSR	Developed	FM	Yes	RD	1989-2003
E139	Upper	Fourmile Creek	USFSR	Reference	FM	Yes	RD	1990-2003

[a] Drainage codes: USFSR – South Fork Salmon River; EFSFSR – East Fork SFSR; LSFSR – Lower SFSR; SR – Secesh River; CHB – Chamberlain Creek; MFSR – Middle Fork Salmon River; MSSR – Mainstem Salmon River; LSR – Little Salmon River; SNR – Snake River.
[b] This reflects the data being collected at these sites at the present time; most sites have additional types of monitoring data as well (*e.g.*, wherever FM data are collected there is likely at least some CE data as well). Codes: SC – Sediment Cores; CE – Cobble Embeddedness; FM – 30-Hoop Free Matrix.
[c] Indicates whether data have been included in most recent core or interstitial reports.
[d] Purpose codes: LRMP – Forest Plan (old LRMP); RD – SFSR Road; MN – Minerals; RG – Range.
[e] Indicates period of record for the data type shown or for the data type with the longest recent record; there are some gaps in most records.
[f] There have been problems with this site and knowing whether the data were collected at the correct location in some years.
[g] Some slight and probably insignificant adjustments in site location over time.

Table 3.—Other sediment monitoring sites on the Payette National Forest with discontinued sampling.

Site Code	Site Name	Stream	Drainage[a]	Status	Data[b]	Report[c]	Purpose[d]	Record[e]
E001	Old South Fork	Blackmare Creek	USFSR	Reference	CE,FM	Yes	LRMP	1989-1997
E003	Old Blackmare	Blackmare Creek	USFSR	Reference	CE,FM	No	LRMP	1989
E004	Lower	SF Blackmare Creek	USFSR	Reference	CE,FM	Yes	LRMP	1989-1999
E027	Oops	SF Blackmare Creek	USFSR	Reference	CE,FM	Yes	LRMP	1990
E065	South Fork	Fourmile Creek	USFSR	Reference	CE,FM	No	LRMP,RD	1989
E066	Mouth	SF Fourmile Creek	USFSR	Reference	CE,FM	No	LRMP,RD	1989
E018	Mouth	SF Buckhorn Creek	USFSR	Developed	CE,FM	No	LRMP	1989
E069	Upper	Fitsum Creek	USFSR	Developed	CE,FM	No	LRMP	1989-1996
E020	Tie Creek	NF Fitsum Creek	USFSR	Developed	CE,FM	No	LRMP	1989
E060	Mouth	Split Creek	SR	Reference	CE,FM	Yes	LRMP	1989-1997
E058	Upper	Lick Creek	SR	Pseudo-Reference	CE,FM	Yes	LRMP	1989-1997
E059	Mouth	NF Lick Creek	SR	Reference	CE,FM	Yes	LRMP	1989-1997
E011	Upper	Cow Creek	SR	Developed	CE,FM	No	LRMP	1989-1997[h]
E012	Middle	Cow Creek	SR	Developed	CE,FM	No	LRMP	1989-1997
E013	Lower	Cow Creek	SR	Developed	CE,FM	No	LRMP	1983[h]
E010	Lower	Maverick Creek	SR	Developed	CE,FM	Yes	LRMP	1989[h]
E141	Upper	Maverick Creek	SR	Developed	CE,FM	Yes	LRMP	1990-1997
E064	Upper	Grouse Creek	SR	Developed	CE,FM	Yes	LRMP	1989-1997
E140	Middle	Grouse Creek	SR	Developed	CE,FM	Yes	LRMP	1990-1997
E072	Lower	Zena Creek	SR	Developed	CE,FM	Yes	LRMP	1983-1997
E074	Upper	Zena Creek	SR	Developed	CE,FM	Yes	LRMP	1989-1997
E075	Mouth	WF Zena Creek	SR	Developed	CE,FM	No	LRMP	1989
E073	Mouth	EF Zena Creek	SR	Developed	CE,FM	No	LRMP	1989-1996
E077	Middle	Threemile Creek	SR	Developed	CE,FM	No	LRMP	1988-1989
E078	Lower	Threemile Creek	SR	Developed	CE,FM	Yes	LRMP	1989-1997
E144	Upper	Porphyry Creek	LSFSR	Reference	CE,FM	Yes	LRMP	1990-1996
E145	Middle	Porphyry Creek	LSFSR	Reference	CE,FM	Yes	LRMP	1990-1997
E045	Upper	Sheep Creek	LSFSR	Reference	CE,FM	Yes	LRMP	1989-1997
E040	Willey Creek	Sheep Creek	LSFSR	Reference	CE,FM	Yes	LRMP	1989-1997
E041	Mouth	Sheep Creek	LSFSR	Reference	CE,FM	Yes	LRMP	1989-1997
E043	South Fork	Sheep Creek	LSFSR	Reference	CE,FM	No	LRMP	1989
E044	South Fork	Sheep Creek	LSFSR	Reference	CE,FM	No	LRMP	1989
E042	North Fork	Sheep Creek	LSFSR	Reference	CE,FM	No	LRMP	1989
E026	Original	Bear Creek	LSFSR	Reference	CE,FM	No	LRMP	1983
E146	Lower	MF Elk Creek	LSFSR	Developed	CE,FM	No	LRMP	1990
E150	Knob Creek	SF Salmon River	LSFSR	Developed	CE,FM	Yes	LRMP	1984-1997
E090	Flats	Jacob's Ladder Creek	MFSR	Reference	CE,FM	No	MN	1990-1997[h]
E121	Mouth	Snowslide Creek	MFSR	Reference	CE,FM	No	MN	1984
E137	West Fork	Monumental Creek	MFSR	Developed	CE,FM	Yes	MN	1984-1997
E147	Holy Terror Creek	Monumental Creek	MFSR	Developed	CE,FM	Yes	MN	1984-1997
E148	Coon Creek	Monumental Creek	MFSR	Developed	CE,FM	Yes	MN	1984-1997
E149	Annie Creek	Monumental Creek	MFSR	Developed	CE,FM	Yes	MN	1984-1997
E089	Culvert	Government Creek	MFSR	Developed	CE,FM	Yes	MN	1984-1997
E091	Trailhead	Smith Creek	MFSR	Developed	CE,FM	No	MN	1990-1997[h]
E092	South Fork	Smith Creek	MFSR	Developed	CE,FM	Yes	MN	1990-1997
E093	Lower	Logan Creek	MFSR	Developed	CE,FM	No	MN	1990-1997[h]
E094	Upper	Logan Creek	MFSR	Developed	CE,FM	No	MN	1990-1997[h]
E095	Mouth	Parks Creek	EFSFSR	Reference	CE,FM	Yes	MN	1994-1997
E049	Mouth	Quartz Creek	EFSFSR	Developed	CE,FM	Yes	MN	1983-1997
E025	Vibika	EF SF Salmon River	EFSFSR	Developed	CE,FM	Yes	MN	1983-1997

[a] Drainage codes: USFSR – South Fork Salmon River; EFSFSR – East Fork SFSR; LSFSR – Lower SFSR; SR – Secesh River; CHB – Chamberlain Creek; MFSR – Middle Fork Salmon River; MSSR – Mainstem Salmon River; LSR – Little Salmon River; SNR – Snake River.
[b] This reflects the data that has been collected at these sites at the present time; most sites have additional types of monitoring data as well. Codes: SC – Sediment Cores; CE – Cobble Embeddedness; FM – 30-Hoop Free Matrix.
[c] Indicates whether data have been included in most recent core or interstitial reports.
[d] Purpose codes: LRMP – Forest Plan (old LRMP); RD – SFSR Road; MN – Minerals; RG – Range.
[e] Indicates period of record for all data collection; most records for any data types will have gaps in the record.
[f] There have been problems with this site and knowing whether the data were collected at the correct location in some years.
[g] Some slight and probably insignificant adjustments in site location over time.
[h] Previously discontinued but sampled in 2004.

LRMP OBJECTIVES AND WATERSHED CONDITION INDICATORS

OLD LRMP

The old LRMP established a general objective of no degradation in sediment conditions for most of the Forest. For the SFSR, however, where historic activities and natural events inundated several important spawning grounds for anadromous fish with fine sediment in the winter and spring of 1964-65, it also established an interim objective for fish habitat in the SFSR of "provid[ing] habitat sufficient to support fishable populations of salmon and trout by 1997." This interim objective was to be interpreted by several means, including primarily the following criteria.

South Fork Salmon River Surface Fines

The old LRMP does not contain specific numeric criteria for surface fines, which were not routinely measured when the old LRMP was published. However, it does indicate that "[p]hotographs should demonstrate that the river is improving as evidenced by characteristics, such as duning and stringing sand, changing from the existing condition toward conditions more similar to those found in Chamberlain Creek, central reaches of the Secesh River, or other appropriate streams" (IV-235). Photographs of several areas were published shortly after the 1964-65 flooding, and additional photographs at standardized locations in the SFSR, including some from the post-flood reports, have been taken approximately annually since 1975 by personnel of the Boise National Forest. There has been no reported effort to compare these photographs with others in either the Secesh River or Chamberlain Creek (though this could probably be done because photographs of monitoring sites are routinely taken), but the SFSR time series photography clearly shows improvement since 1975 (Nelson *et al.* 2004c)[12].

South Fork Salmon River Free Matrix Counts

Free matrix counts are not mentioned in either the old or revised LRMPs, but they have been used on the PNF since 1989 for monitoring interstitial conditions. Free matrix counts are simpler to perform than cobble embeddedness measurements, are likely subject to less observer error, provide a reach-level habitat condition index, and can be used to estimate or predict cobble embeddedness (Nelson *et al.* 1997, 2004d). Because of the apparent usefulness and cost-effectiveness of free matrix counts, Nelson *et al.* (1997) proposed criteria based on this metric[13]:

- Demonstrated improvement in cobble embeddedness or establishment of a statistically significant downward trend using either measured or predicted cobble embeddedness (but not both);

- Measured or predicted embeddedness levels consistently at or near 50% should be considered unacceptable;

and

- Demonstrated improvement in percent free particles from 30-hoop free matrix measurements or establishment of a statistically significant upward trend;

[12] We have posted these images in a screen show format at http://fsweb.payette.r4.fs.fed.us/units/fish.web/FishReports.htm under "Time Series Photography".

[13] There were several typographic and semantic errors in these proposed criteria in Nelson et al. (1997) that we have corrected here.

- Five-year average (or other multi-year average from at least 3 years of data) based on *measured* cobble embeddedness for streams less than 30% embedded at the time standards are adopted should not exceed 30%, and no more than two years out of any five should exceed 35%;

- Five-year average (or other multi-year average from at least 3 years of data) based on *measured* cobble embeddedness for streams more than 30% and less than 40% embedded at the time standards are adopted should not exceed 40%, and no more than two years out of any five should exceed 45%;

or

- Five-year average (or other multi-year average from at least 3 years of data) based on *predicted* cobble embeddedness for streams less than 35% embedded at the time standards are adopted should not exceed 35%, and no more than two years out of any five should exceed 40%;

- Five-year average (or other multi-year average from at least 3 years of data) based on *predicted* cobble embeddedness for streams more than 35% and less than 45% embedded at the time standards are adopted should not exceed 40%, and no more than two years out of any five should exceed 45%.

or

- Five-year average (or other multi-year average from at least 3 years of data) based on *measured* percent free cobbles for streams with more than 20% free at the time standards are adopted should not be less than 20%, and no more than two years out of any five should be less than 15%;

- Five-year average (or other multi-year average from at least 3 years of data) based on *measured* percent free cobbles for streams with more than 10% free and fewer than 20% free at the time standards are adopted should not be less than 15%, and no more than two years out of any five should be less than 10%.

South Fork Salmon River Embeddedness

Under the old LRMP, the SFSR had a specific interim objective of generally improving sediment conditions, but with additional constraints because of the importance of the watershed and the documented degradation in streambed conditions that occurred following the flooding in the winter of 1964-65. Interim sediment standards included (IV-235):

- In locations where cobble embeddedness now exceeds 32 percent, a five-year mean of ≤ 32 percent and no individual year ≥ 37 percent must be observed. Other locations must exhibit no increased sediment deposition outside expected natural variation.

- In locations where percentage fine sediment now exceeds 27 percent, a five-year mean of ≤ 27 percent and no individual year ≥ 29 percent must be observed. Other locations must exhibit no increased sediment deposition outside expected natural variation.

Although these were interim criteria, they were based on studies of local conditions that estimated that embeddedness should naturally range from about 19% to 32% as the proportion of the watershed that was glaciated ranged from 0% to 100%. (Edwards and Burns 1986). The data from which these criteria were derived have expanded since they were proposed as monitoring of sediment conditions has continued as directed by the old LRMP, so we can critically examine the validity of these criteria and, if necessary, propose values that are more reasonable.

South Fork Salmon River Depth Fines

In addition to cobble embeddedness criteria, the old LRMP specified criteria relative to fine sediments smaller than 6.3mm in the gravels of several established spawning areas. These areas had been sampled extensively following the 1964-65 floods, and the old LRMP specified that (IV-236):

- In locations where percentage fine sediment now exceeds 27%, a five year mean of ≤27 percent and no individual year ≥29 percent must be observed.

- Other locations must exhibit no increased sediment deposition outside expected natural variation.

In general, fine sediments have been declining at the upper SFSR spawning areas, and the current average is less than 30% across the board, with exceedences in some years (Nelson *et al*. 2004c). While these generally comply with old LRMP criteria, formal consultation on the SFSR Road Reconstruction Project led to a BO that specifies re-initiation of consultation when intragravel fines exceed specified values at each spawning area (NMFS 1993) Although there have been exceedences, it is unlikely that they are due to the road reconstruction project, which included paving of the road itself, culvert upgrades, closure of many campground roads, and paving of campgrounds that remain open to motor vehicles. In fact, with the occasional exception, potential Chinook and steelhead egg survival, based on the ration of sediments smaller than 9.5mm to sediments smaller than 0.85mm (as suggested by Tappel and Bjornn [1983]) seems to generally be quite good (Nelson *et al.* 2004c).

REVISED LRMP

The revised LRMP does not separate sediment criteria for the SFSR from criteria for the rest of the Forest, but provides default ranges of values by species, and in some cases by life stage, that can be used across the Forest absent better, site-specific information. The default WCI values are to be compared with current habitat conditions and probable post-project conditions in a hydrologic unit (HU) containing a proposed project area to evaluate how the potentially affected watershed is currently functioning and how it will be functioning after project implementation; however, site specific sediment information may be used if it indicates that the default values are inappropriate. The sediment-related WCIs include values for intragravel or surface fine sediments (the "Sediment/Turbidity" WCI) and interstitial fine sediments (the "Substrate Embeddedness" WCI), though evaluated separately here and normally evaluated separately in project planning, are shown together in Table 4 (next page). For the SFSR, we believe that the default values are likely at least somewhat incorrect for the watershed, and NOAA Fisheries Service echoed this concern in their LRMP BO (NMFS 2003).

Table 4.—Default watershed condition indicators (WCIs) from table B-1 of the revised PNF LRMP, expressed as functions of surface or depth fines and cobble embeddedness (Table B-1, pages B-14 and B-15).

Pathways and WCIs	***Functioning Appropriately***	***Functioning at Risk***	***Functioning at Unacceptable Risk***
Sediment/Turbidity (steelhead, chinook)	Low turbidity is indicated by < 12% surface fines (< 0.85 mm)	Moderate turbidity is indicated by 12-20% surface fines (< 0.85 mm)	High turbidity is indicated by > 20% surface fines (< 0.85 mm)
Sediment/Turbidity (in areas of spawning and incubation; rearing areas will be addressed under substrate) (bull trout)	< 12% fines (< 0.85 mm) in gravel. Surface fines (≤6mm) ≤ 20%	12-17% fines (<0.85mm) in gravel. Surface fines (≤ 6mm) are 12-20%.	>17% fines (< 0.85mm) in gravel; Surface fines (< 6mm) or depth fines (< 6mm) > 20% in spawning habitat
Sediment/Turbidity (other fish species: i.e., red band, rainbow, wood river sculpin, etc)	Species-specific criteria should be developed.		
Substrate Embeddedness (Bull trout rearing areas. Spawning and incubation areas are addressed under the Sediment/Turbidity WCI)	Dominant substrate is gravel or cobble (interstitial spaces clear), or embeddedness is < 20%.	Gravel and cobble is subdominant, or if dominant, embeddedness is 20-30%	Bedrock, sand, silt, or small gravel dominant, or if gravel and cobble dominant, embeddedness is > 30%

General Review of WCIs in the Revised LRMP

The ways in which values are presented in the LRMP imply that the effects of fine sediments can be sorted into discrete groups based upon a range of values. While this might make sense from an organizational perspective, we need to point out that there is no biological basis for it. Recent work has demonstrated that increasing sediments have negative effects on salmonid survival at even low levels (Suttle *et al.* 2004), so that even within a range of values considered to be functioning appropriately, increases may, in fact, reduce salmonid survival; this confirms what was previously documented by Chapman and McLeod (1987).

Surface Fines

The origin of the use of measures of surface fines, typically meaning particles smaller than 6.3mm (as in the bull trout "Sediment/Turbidity" WCI is difficult to trace. Chapman and McLeod (1987) do not describe any studies that clearly evaluate the effects of surface fines on salmonids, though it is certainly reasonable that high levels of surface fines would constitute a threat to salmonid survival. There are studies that relate streambed surface composition to salmonids, but these generally talk about embeddedness of large particles by fine sediments or use some estimate of the streambed surface particle composition to compute a substrate score (see *eg.*,Crouse *et al*. 1981; Shepard *et al*. 1984). It seems likely that measuring surface fines was an outgrowth of the popularity of using modifications of Wolman's (1957) pebble count procedure to characterize streambed sediments, despite the fact that the procedure was not designed to measure small streambed particles (Wolman 1954; Bunte and Abt 2001). Pebble counts are not necessary for estimating streambed fines, however, and various types of intersection grids (Bunte and Abt 2001) are also popular and have been suggested and used for various Forest Service monitoring programs (Kershner *et al*. no date; Kershner et al. 2004a). Furthermore, the revised LRMP WCI for "Sediment/Turbidity" relative to anadromous fish is for surface fines smaller than 0.85mm, which is very small for surface estimation methods, and we are not aware of any field studies or monitoring protocols using this size class in surface fines assessments.

We cannot find any studies that directly relate surface fines abundance to salmonids, although many ocular estimation methods for cobble embeddedness (*e.g.*, Bjornn *et al*. 1977; Platts *et al.* 1983) and streambed score incorporate estimates of surface fines composition, and substrate score has been related to salmonid production (Crouse *et al*. 1981; Shepard *et al*. 1984). In fact, Burton *et al*. (1991) suggested visually counting surface fines intersections under an embeddedness hoop as an indicator of interstitial filling by fine sediment. Because of the lack of definitive studies of surface fines and salmonid productivity and the tendency of workers to blur the lines among the different manifestations of fine sediment in and on the streambed, it is difficult to accurately assign criteria for surface fines as an indicator of watershed condition. There are also technical concerns involving high variability in measured surface fines (Nelson *et al*. 2004b; Roper *et al*. 2002), probably related to the mobility of the streambed surface, that reduce their value as indicators of habitat quality for salmonids. The fact that erroneous conclusions with respect to interstitial conditions can result with use of surface fines is documented in Nelson *et al*. (2004b), where results of a field study on several streams on the west side of the PNF and in the chinook and steelhead spawning areas on the east side of the Forest showed high variability in the measurement and poor correlation with intragravel fine sediments (Figure 13).

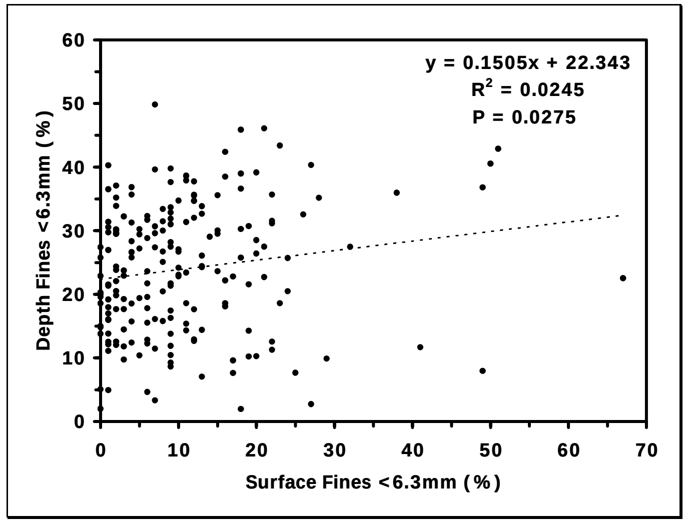

Figure 13.—Regression of depth fines on surface fines, east side sampling (model form is y = bx + a) (from Nelson *et al*. 2004b).

The default values for these WCIs were apparently obtained from a variety of sources, though their applicability to local streams is unclear. For example, surface fines (smaller than 6 mm[14]) must be less than 20% for the system to be in the FA class, but we were unable to verify the source of this number. Overton *et al.* (1995 and 1997) are cited in the WCI table in the LRMP, but they do not show 20% to be an estimate of central tendency of the data from reference (*i.e.*, minimally disturbed) granitic streams, which the former shows to be 23%[15]. In addition, the surface fines WCIs in the LRMP are contradictory in that there is no difference between FA and FR with respect to this indicator[16]. Other examples of our inability to determine the genesis of WCI indices are possible, but it is sufficient to note that the most recent and best information that relates directly to local conditions in the South Fork Salmon River area was not used to establish these criteria.

Cobble Embeddedness

Measurements of cobble embeddedness characterizes the interstitial environment on the streambed surface, which is typically a matrix of free and embedded particles. The interstitial environment is of interest because it provides hiding cover for small fish, including young salmonids, and provides essential habitat for aquatic invertebrates, the principal food source for most stream fish. The influence of the embeddedness of coarse particles by finer particles on stream fish was probably first critically evaluated by Klamt

[14] This size class may be variously identified as 6mm, 6.3mm, 6.4mm, 6.35mm (a direct conversion of the 0.25in to metric; we consider them to be the same and will generally refer to them as "fines smaller than 6.3mm" or "large fines."

[15] The following discussion mentions apparent sources of indicators as indicated by footnotes in Table B-1 of the revised LRMP.

[16] We believe this to be a typographic error and that "≤ 12% was intended; however, this value is not found in Overton *et al.* (1995) from which this WCI was presumably derived.

(1974)[17] and Bjornn *et al.* (1977). Actually measuring embeddedness, however, was first put into practice in the Coast Range of California to estimate the suitability of a small stream for rearing of juvenile steelhead (Kelley and Dettman 1979). The approach was adapted for use on the PNF in the 1980s (Burns 1984; Burns and Edwards 1985) and has been a fundamental component of our sediment monitoring ever since.

Studies investigating the effect of embeddedness on aquatic biota have shown that there are effects that are usually considered undesirable. Bjornn *et al.* (1977) showed deleterious effects to both fish and aquatic insects that are commonly used for food by salmonids. One example is given in Figure 14, which shows the intolerance of the mayfly *Epeorus albertae*, a primary consumer and food source for various aquatic predators, to addition of fine sediment to streambeds of different levels of embeddedness. We have also seen a negative relationship between embeddedness and fish density on the Forest (Figure 15), and some more recent work has shown that growth of juvenile steelhead was increasingly impaired as embeddedness increased (Figure 16, next page) through various mechanisms, including reductions in vulnerable insect prey and increased energy-wasting activity such as swimming and aggression (Suttle *et al.* 2004).

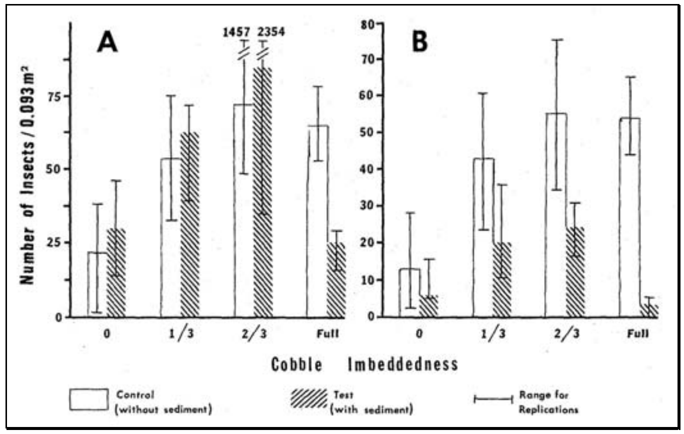

Figure 14.—Differences in insect density in artificial channels in response to four levels of cobble embeddedness. Part A is all insects whereas B is for the mayfly *Epeorus albertae.* (From Bjornn *et al.* 1977).

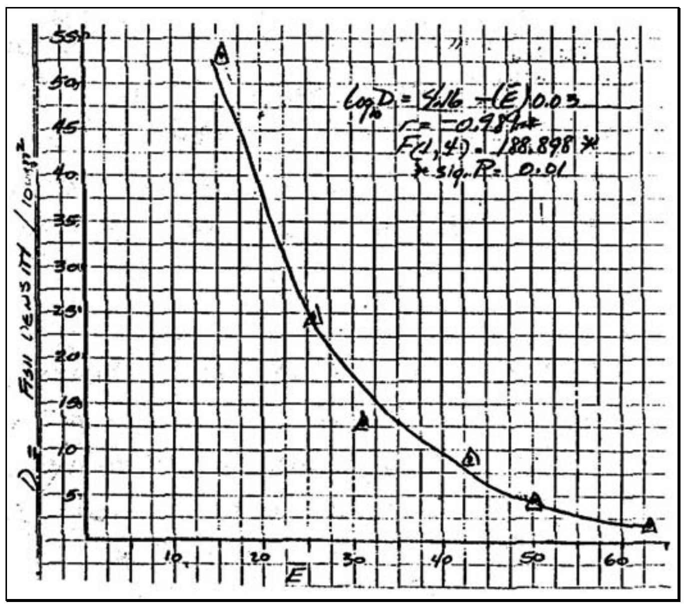

Figure 15.—Relationship of maximum fish density to embeddedness (E) at any individual site (from Chapman and McLeod [1987] citing unpublished work by Russ Thurow and Dave Burns on the Payette National Forest).

Although use of cobble embeddedness measures as an index of habitat quality is relatively widespread and has been specified for use in consultations pursuant to ESA (NMFS 1996) and in state water quality assessments and standards (DEQ 2002), its use as a management indicator has recently been questioned (Sylte 2002; Sylte and Fischenich 2002, 2003). Although many of the concerns

[17] Klamt (1976) seems to cite an M.S. thesis by Prather (1971, *The Effects of Stream Substrate on the Distribution and Abundance of Aquatic Insects*, University of Idaho) as the origin of an streambed embeddedness rating system, and Waters (1995) mentions says that a study by Bjornn and others (1974, *Sediment in Streams and its Effects on Aquatic Life*, University of Idaho) as the first long-term study of the relationships among embeddedness and aquatic ecology, but we have not been able to locate these documents.

expressed by these authors may affect the utility or generality of embeddedness measurements, they demonstrate some unfamiliarity with how we conduct the measurements and analyze the data; they confuse precision of the method of measure with variation in estimates of central tendency[18]. Their criticisms involve primarily technical issues relating to standardization of methods, definitions, variability, and measurement precision, which we ameliorated by use of stratified sampling, training, and continuity of personnel. The potential utility of using cobble embeddedness as an indicator of disturbance in granitic watersheds has been documented (Burns and Edwards 1986; Potyondy 1988), though geology and geomorphology has an influence on what embeddedness levels might be expected under natural conditions (Burns and Edwards 1986). We attempt to minimize the potential for high variability, which is typical of environmental data in general, by our method of stratifying the sampling locations by depth and flow criteria (this stratification is described in most of our interstitial monitoring reports) and requiring that measurements be made after flows have stabilized in the summer.

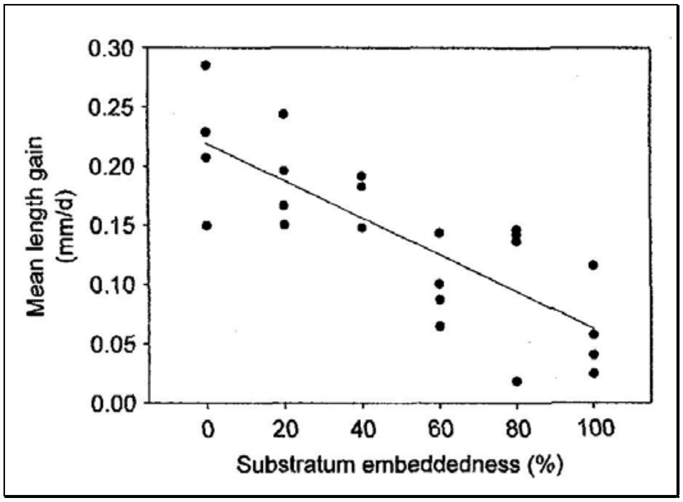

Figure 16.—Relationship between steelhead juvenile growth and substrate embeddedness (from Suttle *et al.* 2004).

The embeddedness standard for FA in the revised LRMP (<20%) would appear, at first glance, to be reasonable. Kelley and Dettman (1979) showed the highest steelhead densities in the range of 15% to about 22% embeddedness and data from Thurow and Burns (unpublished but cited by Chapman and McLeod [1987]) suggested that high densities of salmonids were associated with embeddedness levels below about 30% (Figure 15, previous page). Our own monitoring of embeddedness across the Forest, and in the SFSR in particular, have shown that this level is probably much too low because even sites in undeveloped watersheds have generally had embeddedness levels nearer 30% (Nelson *et al*. 1997 *et sec.*), though highly glaciated watersheds may be somewhat lower naturally and closer to 20% (Burns and Edwards 1986). In addition, the source cited to support the proposed embeddedness WCIs imply that the value refers to embeddedness by particles smaller than 0.85 mm diameter, not by the size particle normally considered in cobble embeddedness measurement in granitic watersheds (particles smaller than 6.3 mm diameter)[19]. Even if the WCI applied to embedding particles smaller than 6.3 mm (large fines), our monitoring of cobble embeddedness in a variety of sites across the Forest indicates that the FA range is generally unachievable even under natural conditions (see Nelson *et al.* 2004d); measurement of embeddedness by particles smaller than 0.85 mm (small fines or silt) has never been attempted on the Forest, but the standard would clearly be even less realistic.

[18] We reviewed draft manuscripts and corresponded with Ms. Sylte on several occasions, and were not convinced that the technique received a correct evaluation.

[19] The revised LRMP cites the watershed analysis manual produced by the Washington Department of Natural Resources (WDNR) in 1993, which is no longer available; however, [we know that] previous WDNR standards specified embeddedness by fines <0.85mm).

Depth Fines

The LRMP WCI values for intragravel fines are similarly confusing. A great deal of study has been directed at salmon and steelhead embryo survival to emergence (STE) in conditions that can be expected to occur in streams of the Idaho Batholith, both in the laboratory and *in situ*, and it is clear that there is an inverse relationship between fine sediment deposition and embryo survival (see Chapman and McLeod [1987] for a review of pertinent studies). However, the WCI identified in the revised LRMP is based on a particle size that is not typical of rock weathering products we normally encounter in the SFSR (*i.e.*, grain sizes in the 4.75 to 6.3mm range) and is not usually featured in survival studies. We do, however, monitor this size fraction in our routine monitoring, there has been some work relating it to STE. In fact, preliminary *in situ* egg survival tests performed in the late 1980s at Poverty Flat, one of the most important of the principal spawning areas in the SFSR, indicated that no chinook embryos survived at intragravel fines smaller than 0.85mm percentages greater than about 6% (Figure 17)[20].

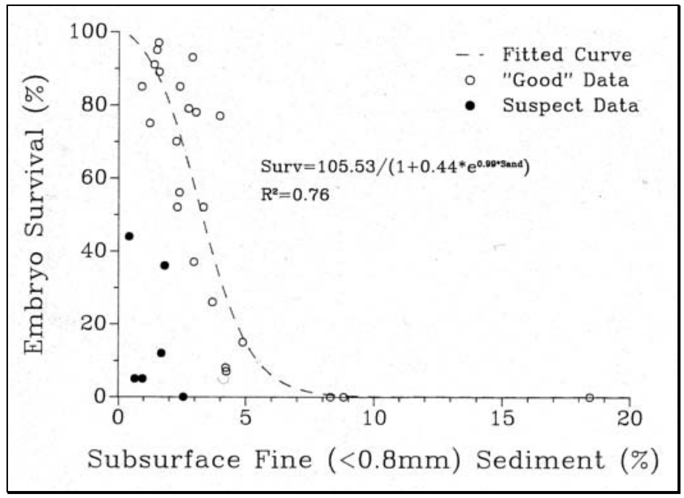

Figure 17.—Observed survival of eyed chinook eggs in egg-baskets *vs.* silt concentration at Poverty Flat, South Fork Salmon River, Idaho (unpublished data from study reported in Nelson and Platts [unpublished] and included in Appendix 3).

No laboratory or field study of salmonid embryo STE can be directly related to intragravel conditions measured in our monitoring program because we cannot incorporate all of the ecological factors controlling embryo survival, and we are generally prohibited from investigating the egg pocket itself. Instead, we try to use what amounts to an index of disturbance related to forest management from which we can estimate whether we are improving or degrading habitat quality for salmonids. Measuring concentrations of fine particles is one of these, but there is some evidence that simply measuring the concentration of fine particles smaller than some specified critical diameter is an inadequate index of intragravel condition (*i.e.*, not only an oversimplification of the ecological factors controlling embryo survival, but an oversimplification of the effects of sediment on embryo survival). The data for Figure 17 are displayed as a nonlinear relationship between fine sediment concentration and survival, but it appears that survival falls off more or less steadily with increasing silt after about 2% intragravel silt. Although the point can be made from Figure 17 that the highest survival values are not seen at the lowest concentrations of silt, it can be made better with displays of similar relationships modeled with large fines (Figure 18, next page) (this figure was developed during the same unpublished research mentioned above, and can be found in Nelson and Platts (no date, attached); other STE studies

[20] This was from *in situ* survival experiments conducted at Poverty Flat during the 1980s by the senior author with the Rocky Mountain Research Station but never published. The "suspect data" identifier in the figure indicates samples where there was an obvious problem (*e.g.*, torn mesh on the egg baskets) that could have affected the final count. Although not completed and published, some of this work was referenced with the methods described in Burton *et al*. (1990); however, the work was incorrectly cited as "Platts and McHenry (unpublished data)" and interpreted the survival data somewhat differently than shown here (note that 70% of the observed points exceed survival predicted by the fitted line in their example).

have shown similar patterns (*e.g.*, McCuddin 1977; Tagart 1984). Furthermore, high survival values can be obtained at sediment concentrations that, on average, produce low or only moderate survival, even in laboratory studies (Figure 19). Use of composite indices, such as geometric mean particle diameter (d_g) or Fredle Index have been used to address this issue, and studies have suggested that they are better indicators of intragravel quality (Young *et al.* 1990); we regularly report d_g. In addition, Tappel and Bjornn (1983) proposed an sediment-based approach to looking at intragravel quality relative to steelhead and Chinook embryo survival that incorporates two grain sizes, 9.3mm and 0.85mm, that we regularly use on the Payette National Forest and may present a better index of the capability of the intragravel environment to promote embryo survival. This method, which we will refer to hereafter as the "pea gravel–silt ratio," will be discussed further in the *Results* and *Discussion* sections.

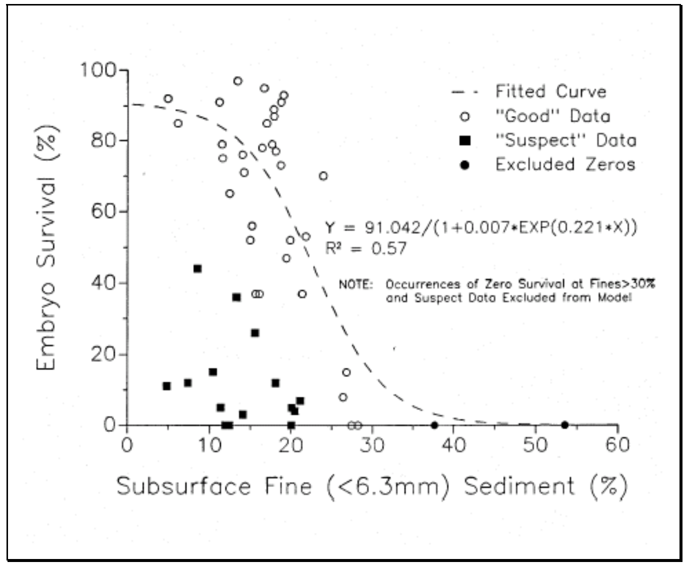

Figure 18.—Observed survival of eyed chinook eggs in egg-baskets *vs.* large fines concentration at Poverty Flat, South Fork Salmon River, Idaho (unpublished data from study reported in Nelson and Platts [unpublished] and included in Appendix 3).

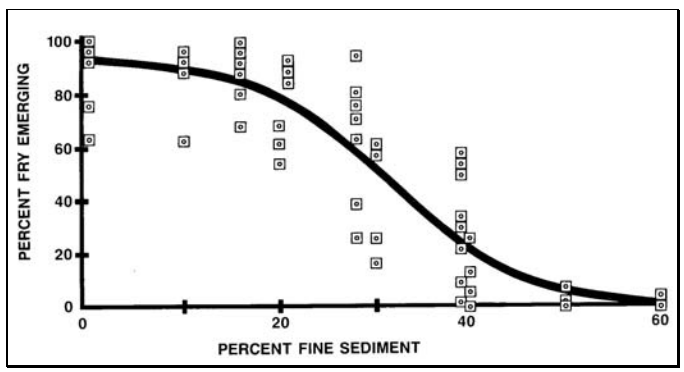

Figure 19.—Relationship of embryo survival and fine sediment (presumably large fines) concentration in laboratory studies (from Stowell *et al.* 1983).

METHODS

STUDY AREAS IN UNDEVELOPED WATERSHEDS

Of the study areas shown in Tables 1 to 3, a fairly large subset consists of sites in watersheds that have experienced little or no timber harvest, mining, or road development; these were identified in the tables as "reference" or "pseudo-reference" sites[21]. The majority of these are in the SFSR watershed (including the East Fork SFSR [EFSFSR], and Secesh River subwatersheds), with a few in the wilderness Chamberlain Creek and Middle Fork Salmon River (MFSR) watersheds (Figure 20, Table 5 [next page]). Most of this area is underlain by the granitic rocks of the Idaho Batholith, but there are large inclusions of Challis volcanics in the Monumental Creek watershed and of metamorphic material in the EFSFSR watershed. This report is principally concerned with establishing sediment indices for granitic watersheds; determination as to whether reference conditions are different among dominant geologies will be accomplished by statistical analyses described below.

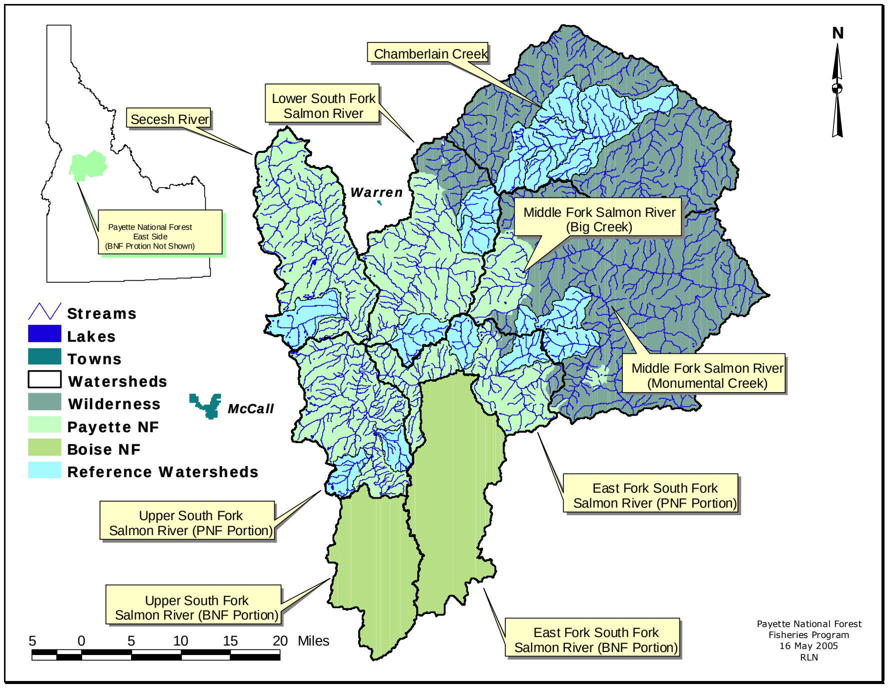

Figure 20.—Study watersheds on the east side of the Payette National Forest.

Site selection as it pertains to this analysis is important to describe here. Sites were established to inventory sediment conditions in various parts of the Forest, especially the SFSR. In contrast to what might be expected in many developed forest areas, developed and undeveloped watersheds in the SFSR are likely to be inherently similar because they are intermingled; consequently, inherent differences between reference and non-reference watersheds as described by Kershner *et al.* (2004b) is unlikely to confound comparisons. The original rationale for selecting sites in the SFSR is described in Burns (1984), when the first sites were selected and we began sediment. A few additional study sites were added to additional developed and undeveloped watersheds as described in Burns and Edwards (1985), Burns (1987), and Ries and Burns (1989), but several new sites were added in 1989 and 1990 to comply with LRMP direction and relevant environmental laws and covering a diversity of watershed conditions (Ries *et al*. 1991). The intent of these additions was to provide a thorough inventory of cobble embeddedness in SFSR tributaries with stream orders from 3 to 5 and to determine time

[21] Sites designated as "pseudo-reference" or "partially developed" have had some management action (such as an access road) but have largely escaped anthropogenic disturbance.

Table 5.—Summary of watersheds used to evaluate reference conditions and for comparisons with conditions representative of developed watersheds.

Catchment	Watershed	Sub-Watershed	Status	Area ac	Area ha	Sites (#)[a]
Upper SFSR	Blackmare Cr	Blackmare Cr	Reference	11,243	4,550	7 (1)
	Fourmile Cr	Fourmile Cr	Reference	9,817	3,973	3 (1)
	Buckhorn Cr	Buckhorn Cr	Non-Reference	16,484	6,671	– (1)
		West Fork	Non-Reference	14,519	5,876	– (0)
		Total		31,003	12,547	– (0)
	Fitsum Cr	Fitsum Cr	Non-Reference	9,425	3,814	– (2)
		NF Fitsum Cr	Non-Reference	10,574	4,279	– (1)
		Total		19,999	8,093	– (3)
	Total			72,062	29,163	10 (6)
Lower SFSR	Sheep Cr	Sheep Cr	Reference	16,262	6,581	7 (0)
	Porphyry Cr	Porphyry Cr	Reference	11,923	4,825	5 (0)
		Wolf Fang Cr	Reference	10,142	4,104	1 (0)
		Total		22,065	8,929	6 (0)
	Pony Cr	Pony Cr	Non-Reference	11,154	4,514	– (1)
	Elk Cr	Lower	Non-Reference	6,631	2,683	– (1)
		Upper	Non-Reference	9,210	3,727	– (0)
		West Fork	Non-Reference	12,099	4,896	– (0)
		Total		27,940	11,306	– (1)
	Total			77,421	31,331	13 (2)
Secesh River	Lick Cr	Lick Cr	Pseudo-Reference	21,825	8,832	4 (1)
	Cow Cr	Cow Cr	Non-Reference	4,742	1,919	– (1)
	Zena Cr	Zena Cr	Non-Reference	7,021	2,841	– (1)
	Grouse Cr	Grouse Cr	Non-Reference	7,565	3,061	– (2)
	Total			41,153	16,654	4 (5)
EFSFSR	Tamarack Cr	Tamarack Cr	Reference	11,716	4,741	1 (–)
	Quartz Cr	Quartz Cr	Reference	12,290	4,974	1 (–)
	Parks Cr	Parks Cr	Reference	4,582	1,854	1 (–)
	Total			28,588	11,569	1 (–)
Chamberlain Cr	Chamberlain Cr	Lower	Reference	10,526	4,260	2 (–)
		Middle	Reference	22,545	9,124	1 (–)
		Flossie-No Name	Reference	11,009	4,455	1 (–)
		Upper	Reference	17,368	7,029	0 (–)
		West Fork	Reference	14,389	5,823	2 (–)
		Moose Cr	Reference	8,579	3,472	0 (–)
		Lodgepole Cr	Reference	11,517	4,661	0 (–)
		Total		95,933	38,824	6 (–)
		McCalla Cr	Reference	26,924	10,896	1 (–)
		Lower Whimstick	Reference	6,390	2,586	0 (–)
		Upper Whimstick	Reference	19,532	7,904	0 (–)
		Total		52,846	21,386	1 (–)
	Total			148,779	60,210	7 (–)
MFSR	Jacobs Ladder Cr	Jacobs Ladder Cr	Reference	NA	NA	1 (–)
	Monumental Cr	Snowslide Cr	Reference	13,381	5,415	1 (–)
		West Fork	Reference	14,243	5,764	1 (–)
		Total		27,624	11,179	2 (–)
	Total			>27,624	>11,179	3 (–)

[a] Shows sites used in subwatershed (6th HU); number in parentheses is number used in reference vs. non-reference comparison; a "0" indicates no sites in subwatershed that is upstream of sites sampled and a "–" indicates not applicable.

trends; thus, though sites were established by stratifying by stream order and the cobble embeddedness depth and flow criteria. Most of the sites used to determine reference conditions were described in Ries *et al*. (1991) as being in undeveloped watersheds; however, a few identified in that report as "partially" developed were used where the effects of Forest development is likely to be unimportant with respect to natural conditions (*e.g.*, in Lick Creek); these sites were thought to be sufficiently undisturbed to illustrate reference conditions and were important for providing an adequate sampling of certain Forest areas. Overall, this collection of sites provides a broad sample of sediment conditions that can be expected to reflect natural conditions, a necessary first step in determining constraints for Forest management.

From a few sampled sites in the mid-1980s to a maximum number of sites sampled in the early nineties, the annual effort has been reduced substantially in recent years. Consequently, individual sites vary in the number of times they have been sampled and their contribution to determination of overall distribution of sediment conditions in time and space is unbalanced. While this may mean that conditions at some sites were weighted too heavily in the analyses, we believe that it is unlikely to compromise the analysis because most sites have several samples and sites in individual watersheds were generally well-distributed. Because of the broad geographic range included, indeed, of the SFSR watershed alone, and because of considerable climatic variation during the monitoring period, we believe that we will be able to get a better idea of expected conditions and temporal variation by using all available reference data despite the unbalanced sampling.

FIELD DATA COLLECTION

Methods for collection of sediment monitoring data have been well documented in our sediment monitoring reports. These reports are available on the PNF's Fisheries Program FSWeb (intranet) page (http://fsweb.payette.r4.fs.fed.us/units/fish.web/FishStart.htm) under "Reports and Publications, Program Reports" and are not reiterated here. These reports not only describe current methods, but also document changes in techniques, if they have occurred, as well. In addition, they also provide detailed documentation of the scope of the sediment monitoring program.

STATISTICAL ANALYSES

Methods for calculating the sediment indices evaluated here are discussed in our sediment monitoring reports and are not reiterated here. We would like to point out, however, that geometric mean particle diameter (d_g) from core sampling can be calculated in several ways, and the results may differ slightly (Shirazi and Seim 1979). We have used the formula for the "Method of Moments" approach provided in Platts *et al*. (1983) with a dry weight correction factor because we actually measure volumes, but we have not assessed the degree to which our values agree with others that have appeared over the years from the same spawning areas. Consequently, these data are useful primarily for comparisons wherein the calculations are performed in the same way and evaluation of trends.

Reference Conditions

Statistical analyses were performed using SAS® 8.01 for Windows®. Simple univariate statistics obtained with PROC UNIVRIATE were used to assess the sediment conditions of reference sites by watershed and by drainage group defined by parent geology (where Group 1 comprises largely granitic watersheds and Group 2 comprises largely metamorphic or volcanic watersheds) and to produce frequency distributions of the individual sediment indices by drainage group. Box and whisker plots were produced using PROC BOXPLOT to display data distributions and means by watershed and drainage group for visual comparison of parameters. Comparisons of surficial and interstitial sediment indices between drainage groups were performed using PROC NPAR1WAY with the "Wilkoxon" option, which produces a nonparametric Wilcoxon rank sum test (statistically equivalent to the Mann-Whitney U-test and analogous to a parametric t-test); tests of core sampling variates among watersheds were also conducted using PROC NPAR1WAY with the "Wilkoxon" option, which, in this case, produces a Kruskall-Wallis test (statistically analogous to parametric analysis of variance). Tests were performed on the means from each site from each sampled year rather than by pooling the entire data set to help normalize the data distributions, although no assumptions of normality have been made. We then used these results to describe the distributions of the sediment data and to provide a framework from which to determine appropriate values for sediment indicators of salmonid habitat condition.

It should be noted that we realize that statistical inferences are weakened somewhat by relying on temporal pseudoreplication (*i.e.*, replication in time) which reduces independence among the samples (Hurlburt 1984), but this seems an acceptable drawback given that pre-project data collected to determine functional category may well have the same problem; often, these data are collected or have been collected to establish the nature of trends in condition, and time is an important component. However, the study sites were well distributed along various environmental gradients (*e.g.*, altitude, latitude, time, etc.), and, as suggested by Oksanen (2001), we have elected to present limited inferential statistics, point out this potential weakness, and allow the reader to decide whether we exceeded the resulting inferential constraints.

Comparison of Reference Sites with Developed Sites

Because the intent of this effort is to suggest more appropriate indicators of watershed condition with respect to salmonid habitat productivity, it is necessary to attempt to evaluate the suitability of the indicators for distinguishing among reference and non-reference conditions. We did this with non-parametric comparisons of means by drainage area and by reference or non-reference status within drainages. These tests were followed by parametric tests using the data as transformed by the most suitable transformation in the reference conditions analysis. For measured embeddedness, the data were from samples collected between 1983 through 1993, for free matrix counts and predicted embeddedness, data were from samples collected from 1989 through 1993, and for surface fines the data were from samples collected from 1991 through 1995. Thus, the samples for each site may not come from the same years, either within data types or between data types, but there is substantial overlap. The measured embeddedness samples were usually not sequential, the free matrix and surface fines samples usually were. We then computed single-year and 5-year mean values (which will also be used to describe revised WCIs) to determine whether there were differences between the two development classes in the SFSR and Secesh River watersheds and to estimate what functional categories would likely result in these streams using the revised indicators. Differences between status class was evaluated using the SAS® nonparametric analysis procedure PROC NPAR1WAY with the "Wilcoxon" option as above. These comparisons are subject to the weakness associated with temporal pseudoreplication as described above. In addition, because means from sites are used instead of data pooled over years, comparisons are made using few pseudoreplicates, but this approach appear to be a reasonable way to investigate the performance of the revised indicators.

Data Quality Issues

Data were drawn from the Payette National Forest Fisheries Program database, a managed relational database that has been under construction since 1994, and interstitial and intragravel monitoring data contains data collected by PNF personnel as well as intragravel data provided by the BNF. The database is being specifically developed to address issues of data quality and integrity of analyses because of historic inconsistencies in data entry proficiency and quality control, electronic data storage method, and analytical methods. In the early stages of development, that is to say, as data were being migrated from various sources (*e.g.*, spreadsheets) to the database, we calculated an error rate in the cobble embeddedness data (which we assume to be representative of other our types of sediment data as well) of approximately 5%. Data are now entered directly, and because of error checking and validation measures that we have established for the data entry process, the error rate is now probably much less; however, we have not attempted to estimate it.

We are not aware of data collection problems with either free matrix or surface fines data, but in the mid-1990s, we became aware of problems with some of our cobble embeddedness data that we believe resulted from improper field measurements. These problems were manifested as an inability to establish an adequate linear regression of embeddedness on free matrix. This relationship is normally highly significant ($P < 0.01$) with a coefficient of determination (r^2) of about 0.30 or larger (Ries and Burns 1989; Nelson *et al*. 1996 *et seq*.). We first documented this problem in 1996 (Nelson *et al*. 1996), which we attributed to insufficient training and poor field collection methods and attempted to remedy with more intensive pre-season training sessions; more recent results (Nelson *et al.* 2004d) suggest that we were correct because we have re-established consistently acceptable linear relationships. Unfortunately, however, several years were identified as having potentially erroneous cobble embeddedness data because of this, though no attempt has been made to determine which sites were likely sampled correctly and which were not. Consequently, time series analyses of cobble embeddedness trends excluded these potentially inappropriate samples. This concept was followed here, wherein cobble embeddedness data from 1992, 1994, 1995, and 2000 were excluded from the statistical analyses.

Results

As mentioned previously, sediment monitoring results have been updated and reported frequently; all of the reports have been posted on the Fisheries Program FSWEB location. The following discussion presents a detailed look at the overall conclusions supported by these reports and some updated analysis as needed.

Surface Fines

We have monitored surface fines on the Payette National Forest, primarily in granitic watersheds, since 1991. Evaluation of the data appeared in Nelson *et al.* (2003b), which did not evaluate surface fines at all reference sites shown in Tables 1 to 3, nor did it report on the average condition for the reference sites analyzed, but summarized data are presented in the appendices of Nelson *et al.* (2004d); however, looking through the values presented in the Nelson *et al.* (2003b), it seems that surface fines for reference sites average about 10 to 20% with sites in developed areas sometimes, but not necessarily, significantly higher in surface fines. This does not seem to conflict with the revised LRMP WCI for fines smaller than 6mm in bull trout spawning and incubation areas, but the anadromous fish criteria of up to 12% surface fines smaller than 0.85mm for an FA rating is too high (although we know of no studies documenting surface fines relative to this particle size class).

Reference Conditions

Visual inspection of the surface fines data distributions by watershed and drainage group suggest that conditions vary somewhat among granitic watersheds but that they are generally similar, whereas conditions in the other watersheds appear to have generally fewer fines and less variability (Figure 21); both are positively skewed (Figure 22, next page), so the median is a better estimator for the centers of the distributions. The nonparametric test for differences among drainage groups was significant ($P < 0.0001$, 1 df), and we conclude that granitic watersheds have, on average, more surface fines than the other watersheds and a broader inherent range of values. There were some obvious differences among the granitic watersheds, specifically that the lower SFSR (LSFSR) appeared to have somewhat lower mean fines[22] and the upper SFSR (USFSR) appeared too have somewhat higher mean fines with greater variability, but we believe that grouping them for determination of natural conditions and the distribution of the surface fines data is reasonable.

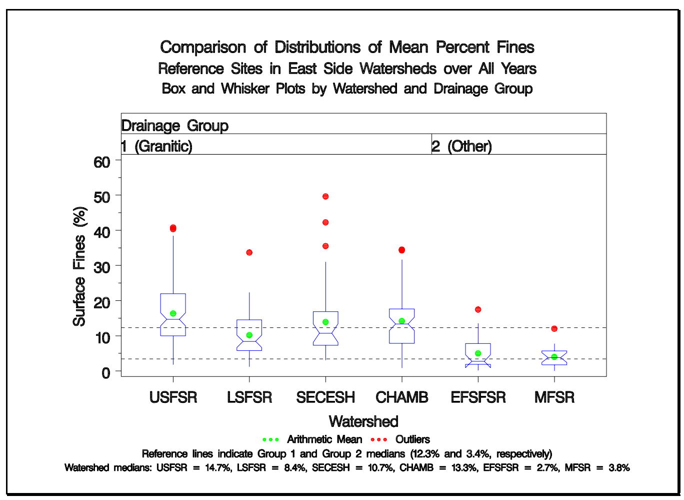

Figure 21.—Box and whisker plots comparing distributions of surface fines means from reference sites by watershed and drainage group.

[22] In fact, we know that one stream in the LFSFSR, Porphyry Creek, is atypical of granitic streams in this respect (Nelson *et al.* 1997 *et seq.*); without Porphyry Creek, the LSFSR streams would likely be more like the other streams in this group.

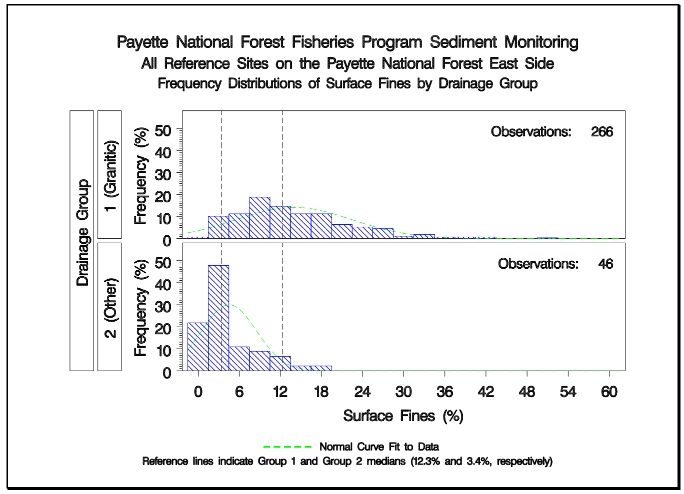

Figure 22.—Observed frequency distribution of reference site surface fines means by drainage group.

Specifics of the distributions of surface fines means by drainage group (Table 6) show just how much different these areas are, with 95% of the values in Group 2 being equal to the median (50th percentile) value in Group 1. In both groups, the medians are smaller than the means. We have questioned the use of mean or median surface fines on their own for determining salmonid habitat quality, because streambed conditions are subject to a great deal of temporal variability and we've not been able to establish clear relationships between surface and subsurface conditions (Nelson *et al.* 1996, 1997,2004b), although the coefficient of variation (CV) here was less than that reported in Nelson et al. (2004b). This is illustrated for selected reference sites in Figure 23, which shows annual variation in relation to the percentiles shown in Table 7[23]. Several things stand out pretty clearly in this graph:

- The temporal variation for the Blackmare Creek site (USFSR) seems to be higher than the others, but it was within the interquartile range about[24] 69% of the time.
- The Chamberlain Creek site (CHAMB) was within the interquartile range about half the time.

Table 6.—Selected percentiles derived from the distribution of mean percent surface fines in reference sites (interquartile range is indicated by double lines).

Drainage Group	Mean	CV (%)	Per-centile	Value
Group 1	14.0	60.3	0	0.8
			5	3.3
			10	4.4
			20	7.2
			25	8.1
			50	12.3
			75	18.4
			80	19.9
			90	25.5
			95	29.2
			100	49.6
Group 2	4.5	88.4	0	0.0
			5	0.1
			10	1.0
			20	1.4
			25	1.8
			50	3.4
			75	6.3
			80	7.0
			90	11.3
			95	12.3
			100	17.4

- The median (50th percentile) is met or

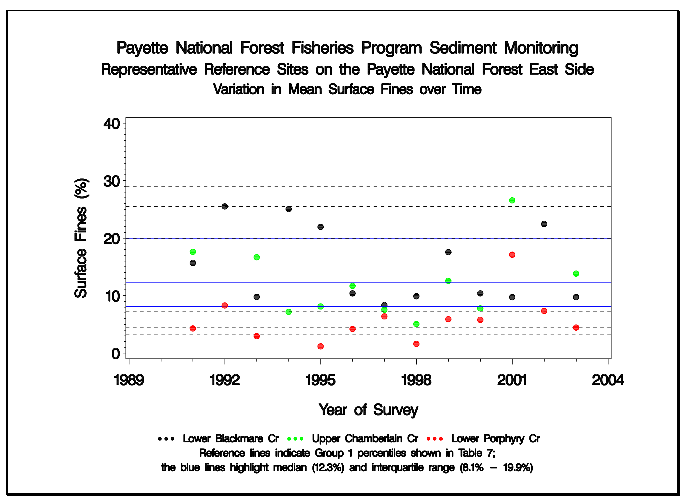

Figure 23.—Annual variation in surface fines for selected reference sites relative to percentiles determined from univariate analysis of the site means.

---

[23] Although the mean is not shown in Figure 7, the median (50th percentile) is shown; both are estimators of central tendency and would be equivalent in a perfectly normal distribution.

exceeded by all sites about half the time as we would expect.

- The Blackmare Creek and Chamberlain Creek sites both met or exceeded the 75th percentile (19.5%) at least once; the USFSR site exceeded it 4 times.

- The Porphyry Creek site (LFSFSR) typically, but not always, had fewer surface fines than the other sites and exceeded the 50th percentile, or median value, of 12.8% only once.

- The Chamberlain Creek site was typically intermediate in surface fines between the other two sites.

These sites were selected for illustration without any particular bias in an effort to display one site from the three SFSR areas. Other sites could have been used and specific relationships among them may be different, but these seem likely to be representative.

Reference — Non-Reference Comparisons

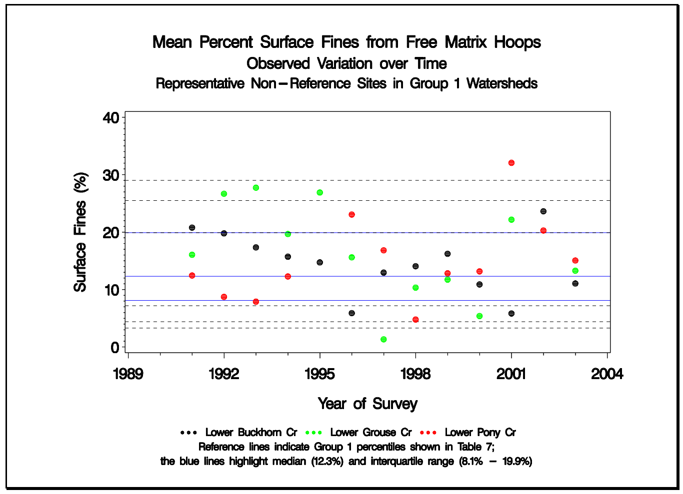

Figure 24.—Annual variation in surface fines for selected non-reference sites relative to percentiles determined from univariate analysis of the site means.

Figure 24 displays the mean surface fines levels for several sites in developed portions of the Forest that are geographically near their reference stream counterparts (same colors in Figures 23 and 24 indicate similar geographic positions, except that no developed site was available in Chamberlain Basin, so Grouse Creek, was selected for display). Several additional statements can be made by visually comparing these two figures:

- The developed sites appear to vary more around the median, though this does not seem true of the USFSR sites (Blackmare Creek and Buckhorn Creek).

- The median (50th percentile) is met or exceeded by all of these sites more than half the time.

- The Pony Creek site is more similar to the USFSR sites than to the LSFSR Porphyry Creek, which is because of a difference in lithology in the Porphyry Creek watershed (Nelson *et al.* 1997 *et seq*.).

- The conditions at the Grouse Creek site (SECESH) were within the interquartile range about 42% of the time.

- The Buckhorn Creek site (USFSR) was within the interquartile range about 62% of the time.

[24] These bulleted statements use "about" because they are generated visually from the graphics rather than from the numeric data analysis, and markers relationships to the reference lines are sometimes obscured.

- The Pony Creek site appeared to have the highest variability, but was still within the interquartile range about 55% of the time.

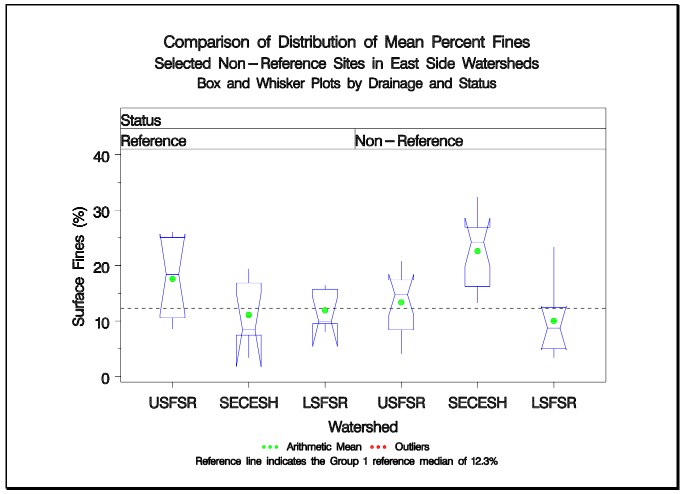

Figure 25.—Box and whisker plots comparing mean surface fines data distributions in selected SFSR sites by watershed and status.

The data displayed in Figure 24 suggest that surface fines in the non-reference sites may not be much higher than in the reference sites, but that was for representative non-reference streams, not for streams known to have had sediment deposition problems in the 1960s. These streams were concentrated in the lower Secesh River and USFSR watersheds; the distributional comparison (Figure 25) suggests that surface fines remain higher in these watersheds (the LSFSR is shown for comparison). The nonparametric Kruskall-Wallis test does confirm significant differences in the non-reference streams among these watersheds (P = 0.0002, 2 df), but it does not reveal the precise nature of the differences. Comparisons by watershed are investigated further below.

Upper South Fork Salmon River

The USFSR watershed was severely affected by hillslope failures caused by the 1964-65 Christmas storms and an extensive logging road network. Nonparametric tests indicated that there were detectable differences among the sites (P = 0.0473, 4 df), but there was no clear pattern because the highest fines value in this comparison came from a reference site (Table 7) and the two sites with the lowest fines have had some development (the two Fitsum Creek sites, which are adjacent to some lower SECESH sites that were severely affected). The fact that these data come from 1991 through 1995, nearly two decades after the landsliding occurred, may indicate that streambed surface conditions have recovered.

Table 7.—Differences in surface fines among sampling sites in the USFSR watershed.

Stream	Status	Sample Size (N)	Surface Fines (%)
Blackmare Creek	Reference	5	19.6
Fourmile Creek	Reference	5	15.6
Buckhorn Creek	Non-Reference	5	17.7
Fitsum Creek	Non-Reference	5	13.8
Fitsum Creek	Non-Reference	5	8.6

Secesh River

Statistical testing also indicated differences among watersheds in the SECESH watershed (P = 0.0065, 3 df), but here it does seem that some of it is due to differences based on development status because the lowest value was from a reference site and the highest was from a harvested watershed (Table 8).

Table 8.—Differences in surface fines among sampling sites in the SECESH watershed.

Stream	Status	Sample Size (N)	Surface Fines (%)
Lick Creek	Reference	5	11.1
Grouse Creek	Non-Reference	5	23.4
Cow Creek	Non-Reference	5	30.0
Zena Creek	Non-Reference	5	16.4

FREE MATRIX COUNTS

Free matrix counts are not addressed in the revised LRMP, but they represent another important method for assessing interstitial conditions in the streambed and are correlated with embeddedness. In fact, we have proposed them previously for use instead of cobble embeddedness measurements because of technical problems in the latter and their usefulness for estimating cobble embeddedness (Nelson *et al*. 1997 *et seq.*). We are evaluating free matrix counts from reference watersheds here because we anticipate a high likelihood that any modified WCIs we propose would use this index of fish habitat condition.

Reference Conditions

Differences in amount of free particles clearly existed among watersheds (Figure 26), with the granitic watersheds having fewer free particles, on average, than the Group 2 watersheds (Kruskall-Wallis P < 0.0001, 1 df). The Group 1 data distribution also appeared to be more highly skewed than the Group 2 data dsitribution, with the positive skew (Figure 27) shifting the mean more to the right (*i.e.*, toward more free particles) than the median (Table 9).

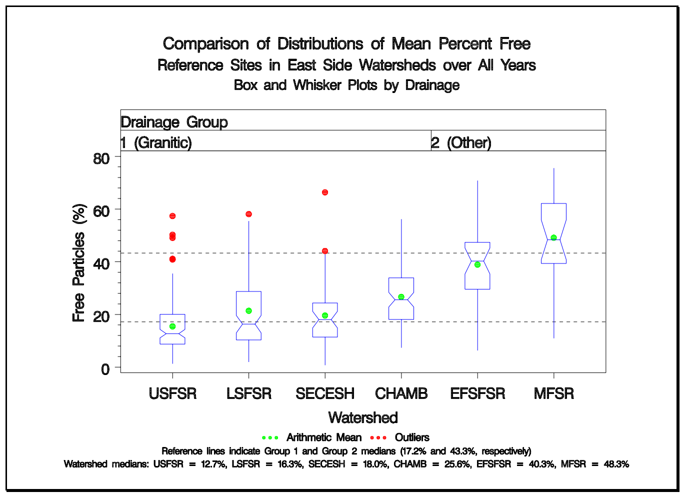

Figure 26.—Box and whisker plots comparing free matrix distributions from reference sites by watershed.

Inspection of the annual data for selected reference sites and the pattern of variation shown in Figure 28 (next page) reveals the following:

- The LSFSR site on Porphyry Creek site almost invariably had more free particles than the sites in the other two watersheds.

Table 9.—Selected percentiles derived from the distribution of mean percent free matrix in reference sites (interquartile range is indicated by double lines).

Drainage Group	Mean	CV (%)	Per-centile	Value
Group 1	20.2	62.0	0	0.7
			5	5.0
			10	7.2
			20	9.7
			25	10.5
			50	17.2
			75	26.6
			80	28.9
			90	36.4
			95	48.2
			100	66.4
Group 2	43.2	36.8	0	6.3
			5	16.8
			10	21.8
			20	29.6
			25	33.6
			50	43.3
			75	53.6
			80	55.5
			90	64.8
			95	68.0
			100	75.5

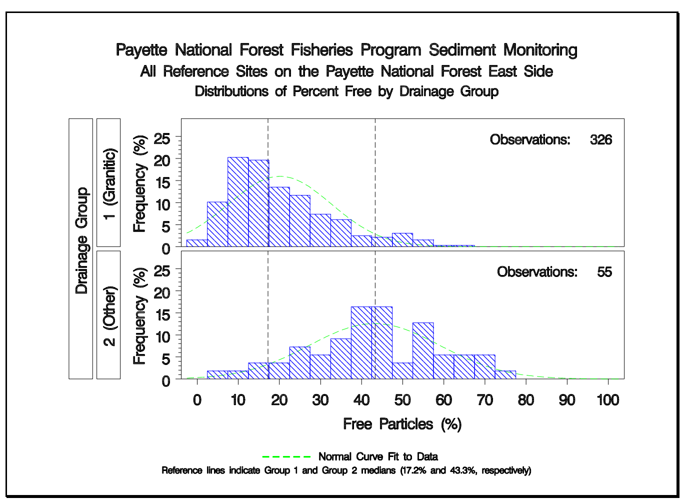

Figure 27.—Observed frequency distributions of reference site free matrix means by drainage group.

- The USFSR site on Blackmare Creek typically had fewer free particles than the site on Chamberlain Creek.

- About 73% of the time the USFSR site was at or below the median Group 1 value (17.2%) and below the 25th percentile (10.5%) 40% of the time.

- The Chamberlain Creek site was within the Group 1 interquartile range approximately 75% of the time, and was below the 25th percentile only once.

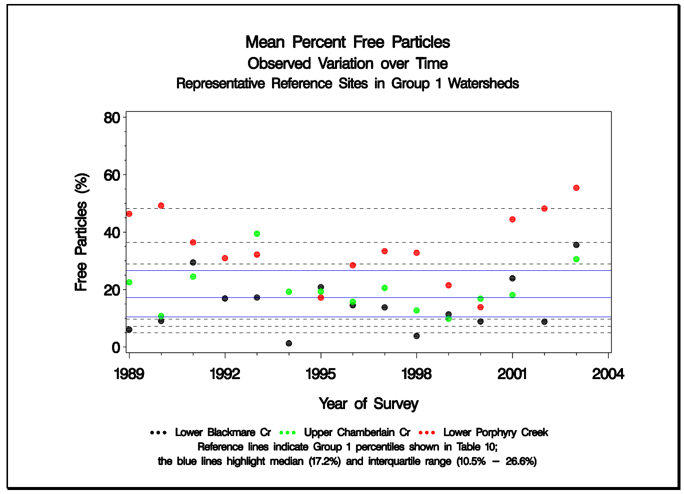

Figure 28.—Annual variation in mean free matrix for selected reference sites relative to percentiles determined from square root transformation of the site means.

- The Porphyry Creek site (LSFSR) was at or below the Group 1 median only once and appeared to have the highest variability.

Reference — Non-Reference Comparisons

Non-parametric comparison of free matrix means from SFSR and SECESH reference and developed sites (not shown) indicated that percent free particles was not significantly different between reference and developed areas, though the parametric t-test suggested that there might be slightly higher free matrix in the developed sites. Conditions in the selected sites in developed Forest watersheds are displayed in Figure 29. Inspection of this figure leads to the following generalizations:

- The Buckhorn Creek (USFSR) site typically had fewer free particles than either of the other two sites and was below the 25th percentile (10.5%) about half the time and never exceeded the 75th percentile (26.6%).

- The Buckhorn Creek site was below the median (17.2%) 93% of the time.

- The Grouse Creek site (SECESH) was at or above the median 77% of the time and was at or below the 25th percentile only once.

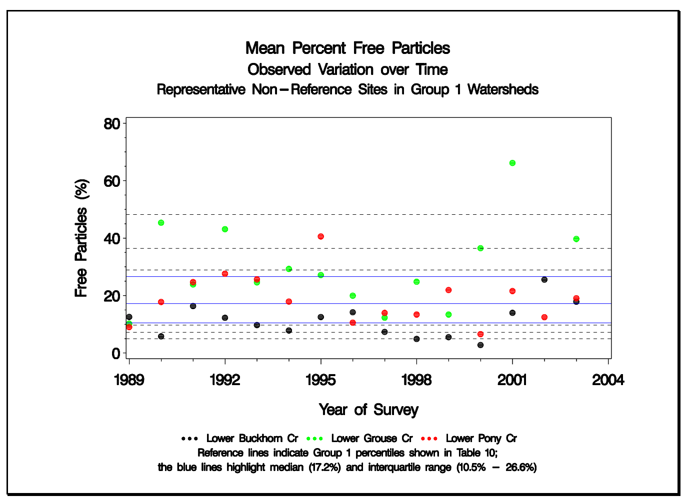

Figure 29.—Annual variation in mean free matrix for selected non-reference sites relative to percentiles determined from square root transformation of the site means.

- The Pony Creek site (LSFSR) was within the interquartile range about 73% of the time but was below the 25th percentile only twice.

- The Grouse Creek site seemed to have the most variability; the Buckhorn Creek site the least.

Free matrix conditions were not demonstrably different for non-reference streams among watersheds (Kruskall-Wallis P = 0.7155, 2 df), but the distributional comparison (Figure 30) suggests that this might be the result of the SECESH reference site used for this comparison having a average higher free matrix count (26.6%) than the reference median (17.2%) during this period and the relatively low power of the test.

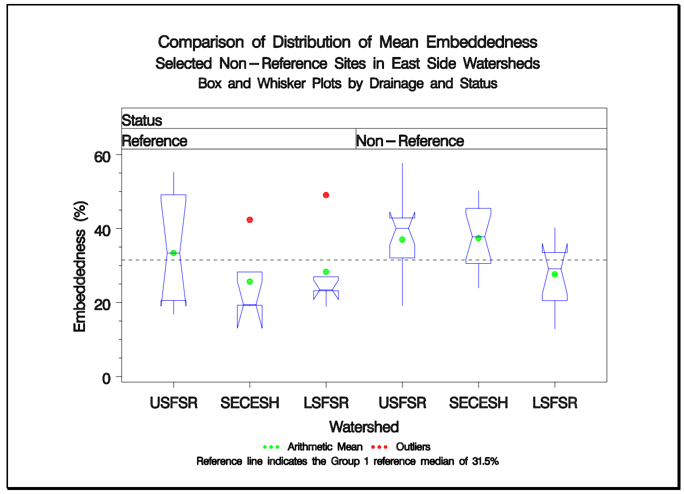

Figure 30.—Box and whisker plots comparing free matrix distributions from selected sites in the SFSR by watershed and status.

Upper South Fork Salmon River

Inspection of Figure 30 suggested the possibility of differences in the USFSR among sites during the test period, but the Kruskall-Wallis test revealed no differences (P = 0.1075, 4 df), though the test is relatively weak and the P-value is near the 10% probability level we use for determining significance. However, inspection of the data (Table 10) does not suggest actual differences based on reference status, because one of the lower values is from a reference site (Blackmare Creek) while one of the higher values is from a non-reference site (Fitsum Creek). These watersheds were not as affected by the 1960s floods as those in the lower SECESH watershed, and apparently, measurable change in free particles occurred after the floods, it is either no longer detectable or was similar in the reference watersheds.

Table 10.—Differences in surface fines among sampling sites in the USFSR watershed.

Stream	Status	Sample Size (N)	Free Particles (%)
Blackmare Creek	Reference	5	14.2
Fourmile Creek	Reference	5	21.3
Buckhorn Creek	Non-Reference	5	10.9
Fitsum Creek	Non-Reference	5	20.7
Fitsum Creek	Non-Reference	5	15.8

Secesh River

The distributions shown in Figure 30 also suggest differences between the SECESH reference and non-reference sites during the test period, but, again, the Kruskall-Wallis test failed to detect them (P = 0.1075, 3 df). This is likely the result of the Grouse Creek site having high average free matrix counts during this period (27.4%) and the Zena Creek and Cow Creek sites having lower values (Table 11). The Grouse Creek site was not in the area severely affected by the 1960s floods, whereas both the Zena Creek and Cow Creek sites were developed and severely affected by them; however, both of these low values were within the interquartile range for reference sites, suggesting considerable improvement in the more than two decades that have elapsed since the floods.

Table 11.—Differences in surface fines among sampling sites in the SECESH watershed.

Stream	Status	Sample Size (N)	Free Particles (%)
Lick Creek	Reference	5	26.6
Grouse Creek	Non-Reference	5	27.4
Cow Creek	Non-Reference	5	16.8
Zena Creek	Non-Reference	5	13.0

MEASURED COBBLE EMBEDDEDNESS

Interstitial monitoring, as performed on the PNF, includes both cobble embeddedness measurements and free matrix counts; one or the other has been performed consistently at a large number of sediment sites since 1984. Cobble embeddedness, which measures the degree to which surface cobbles have a gasket of fine sediments, was developed for granitic watersheds in California by Kelley and Dettman (1979) and was formalized for similar geology in the Idaho Batholith here on the PNF (Burns 1984; Burns and Edwards 1985). It has also been used elsewhere and is more widely used and understood outside the PNF than free matrix counts; thus, it was a reasonable selection for a WCI, though recent monitoring has suggested that free matrix counts might be better (Nelson *et al.* 1997 *et seq.*). Measurement of cobble embeddedness requires relatively precise measurements and consistent application of depth and flow criteria (Burns and Edwards 1985); this can be problematic and, we believe, lead to suspect results if specified procedures are not followed closely (Nelson *et al*. 1997 *et seq.*).

The revised LRMP embeddedness considers values less than 20% embedded to be "functioning appropriately."[25] Previous analyses (Nelson *et al*. 1997) suggested that even the more liberal standards in the old LRMP were likely too stringent because two sites draining the undeveloped SFSR watersheds of Blackmare Creek and Fourmile analyzed in that report failed to attain the values; Nelson *et al*. (2004d) show these sites exceeding the 20% level about 25-30% of the time. This is assuming that the new LRMP value relates to fine sediments of 6.3mm and smaller, as we commonly understand embeddedness.

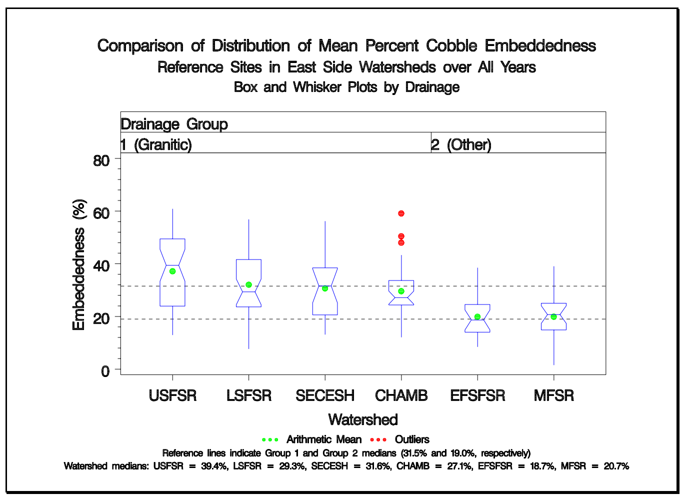

Figure 31.—Box and whisker plots comparing mean measured cobble embeddedness distributions from reference sites by watershed and drainage group.

Reference Conditions

Because of the problems with directly measuring cobble embeddedness mentioned above, we have reduced our embeddedness sampling effort since 1997 in favor of free matrix counts. Because embeddedness can be calculated from free matrix counts, however, we do maintain a sufficient cobble embeddedness sample each year to allow double sampling for embeddedness calculation and to serve as a quality control measure. We can use these data to assess reference conditions, but the data from non-granitic watersheds is very limited. Non-parametric testing of embeddedness between drainage groups was significant ($P < 0.0001$, 1 df), with the Group 1 median higher than that of Group 2 (Figure 31).

The data sets from Group 1 and Group 2 had somewhat similar distributions, with the latter shifted toward lower levels of embeddedness (Figure 32, next page). Because

[25] As written, the WCI appears to apply only to bull trout rearing areas, but this is an incorrect interpretation as it was intended to apply to all species and the parenthetical indication for bull trout was an attempt to point at the USFWS bull trout matrix (W.M. Lind, Fishery Biologist, NOAA Fisheries Service, Boise, Idaho, personal communication).

Table 12.—Selected percentiles derived from the distribution of mean percent cobble embeddedness measured in reference sites (interquartile range is indicated by double lines).

Drainage Group	Mean	CV (%)	Per-centile	Value
Group 1	32.9	36.9	0	7.6
			5	15.5
			10	18.8
			20	22.3
			25	23.9
			50	31.5
			75	41.6
			80	43.8
			90	50.4
			95	54.5
			100	60.8
Group 2	19.9	38.3	0	1.5
			5	11.3
			10	12.2
			20	13.7
			25	14.2
			50	19.0
			75	24.5
			80	25.0
			90	29.3
			95	32.9
			100	39.0

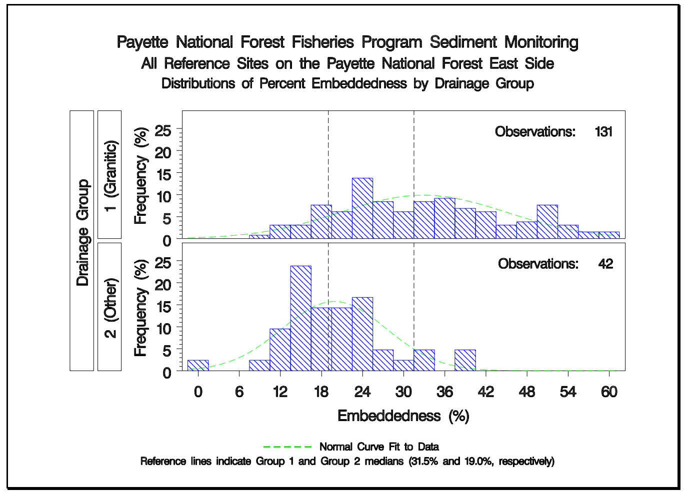

Figure 32.—Observed frequency distribution of measured reference site cobble embeddedness means by drainage group.

there is some skew in the Group 1 distribution, the median of 31.5% (Table 12) is probably a better estimate of the center of the distribution than the mean (32.9%).

Although the coefficients of variation in both groups were relatively low, annual variation for individual sites was substantial (Figure 33[26]). The following points relative to this variation can be made:

- The Blackmare Creek (USFSR) and Chamberlain Creek sites were typically more highly embedded than the Porphyry Creek site (LFSFSR).
- All of the sites regularly met or exceeded the 20% limit in the revised LRMP, and the USFSR site was below it only once.
- Both the USFSR and Chamberlain Creek sites exceeded the 75th percentile (41.6%) at least once.
- The Blackmare Creek site met or exceeded the median (31.5%) 82% of the time but was within or below the interquartile range (23.9% to 41.6%) about half of the time.
- The Porphyry Creek site met or exceeded the median value about 40% of the time.

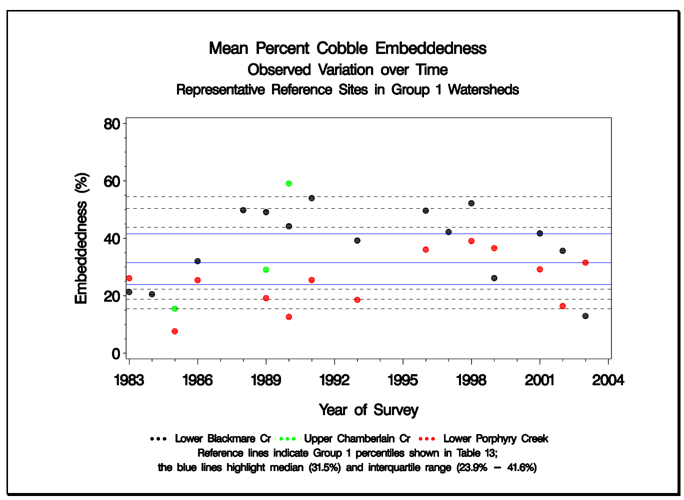

Figure 33.—Annual variation in measured cobble embeddedness for selected reference sites relative to percentiles determined from univariate analysis of the site means.

[26] Note that the Chamberlain Creek site has a short sampling record because of cutbacks in monitoring.

Reference — Non-Reference Comparisons

As with the reference sites, annual variability was generally high in selected non-reference sites (Figure 34); it is interesting to note, however, that the variation at the Buckhorn Creek site appeared to be much smaller than at the site in the adjacent Blackmare Creek. From the graph in Figure 34, the following generalizations can be made:

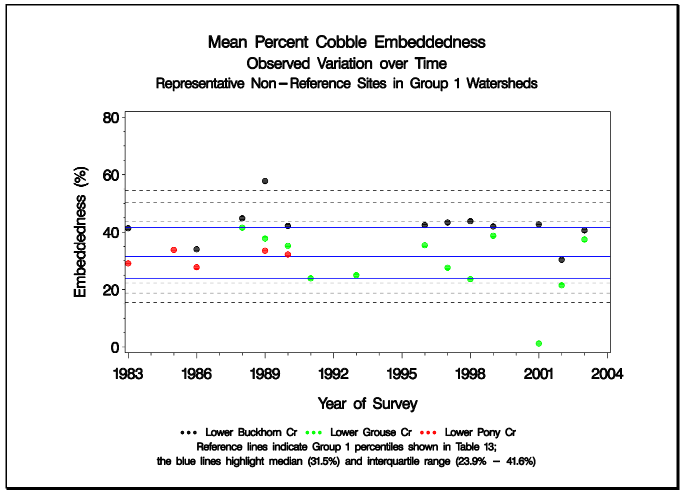

Figure 34.—Annual variation in measured cobble embeddedness for selected non-reference sites relative to percentiles determined from univariate analysis of the site means.

- Only the site on Grouse Creek (SECESH) had values below the 25th percentile (22.3%).
- The Grouse Creek site was within the interquartile range (23.9% to 41.6%) 64% of the time.
- The site on Buckhorn Creek (USFSR) was within the interquartile range about 22% of the time and exceeded the 75th percentile consistently before 2002.
- The site on Buckhorn Creek also exceeded the median (31.8%) about 83% of the time.
- Variability at the Grouse Creek site appeared to be very high, but the 2001 value seems to be too low.
- The Pony Creek site (LFSFSR) had embeddedness levels very near, but slightly above, the median in every sample, but sampling was insufficient to paint a good picture of variability there.

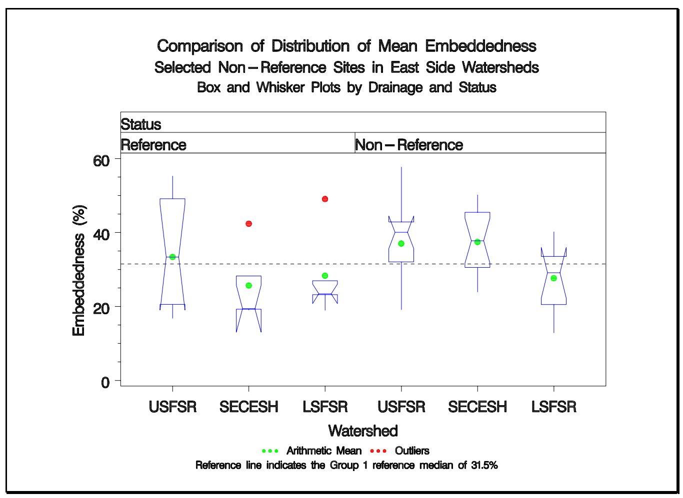

Figure 35.—Observed frequency distribution cobble embeddedness distributions from selected sites in the SFSR by watershed and status.

The distributional comparison by watershed and status (Figure 35) suggests that embeddedness generally remained elevated for some time after the flood events of the mid-1960s in the lower Secesh and upper SFSR watersheds, and the nonparametric Kruskall-Wallis test does confirm significant differences in the non-reference streams existed among these watersheds (P = 0.0312, 2 df). The precise nature of the differences is not revealed by the test, though it seems clear that embeddedness was, on average, lower for the LSFSR than for the other two watersheds.

Upper South Fork Salmon River

Nonparametric tests indicated that there were no detectable differences among the sites (P = 0.4656, 4 df), though it did appear that the Buckhorn Creek site was higher than the others (Table 13), which were all very similar. Most values were very near the reference median from the previous analysis, but the Buckhorn Creek site averaged higher than the 75th percentile during this sampling period (all samples were collected before 1990).

Table 13.—Differences in embeddedness among sampling sites in the USFSR watershed.

Stream	Status	Sample Size (N)	Embeddedness (%)
Blackmare Creek	Reference	5	34.6
Fourmile Creek	Reference	5	32.2
Buckhorn Creek	Non-Reference	5	44.0
Fitsum Creek	Non-Reference	5	32.0
Fitsum Creek	Non-Reference	5	32.9

Secesh River

No statistically significant differences in embeddedness among SECESH watershed sites were revealed by the nonparametric Kruskall-Wallis test (P = 0.1130, 4 df), though the P-value was very near significance level (0.10) we would use to reject the null hypothesis of no difference. It is clear that the Cow Creek and Zena Creek sites, which were heavily impacted in the mid-1960s, appeared to have very high embeddedness levels even as late as the late-1980s (Table 14), despite the inability of this rather weak nonparametric test to reveal differences, with 5-year average values near the 75th percentile from the reference analysis.

Table 14.—Differences in surface fines among sampling sites in the SECESH watershed.

Stream	Status	Sample Size (N)	Embeddedness (%)
Lick Creek	Reference	5	25.6
Grouse Creek	Non-Reference	5	33.4
Cow Creek	Non-Reference	5	38.3
Zena Creek	Non-Reference	5	40.6

PREDICTED COBBLE EMBEDDEDNESS

The PNF has been promoting the concept of predicting cobble embeddedness from free matrix counts for three principal reasons: (1) free matrix monitoring is a simpler process with less chance for sampling error, (2) it is reach-level sampling rather than habitat-specific sampling, and (3) double sampling can serve as a quality control mechanism for the cobble embeddedness measurements. The regression models built to predict embeddedness from free matrix cannot be expected to yield the same numbers as embeddedness sampling, however, because there is always substantial unexplained variation in them. Consequently, it is useful to examine the distributions of predicted cobble embeddedness in reference sites; these are summarized in Table 15. This shows that the median predicted embeddedness is about four percentage points higher than the corresponding measured value in both drainage groups; therefore, any proposed

Table 15.—Selected percentiles derived from the distribution of mean percent cobble embeddedness predicted from free matrix counts in reference sites (interquartile range is indicated by double lines).

Drainage Group	Mean	CV (%)	Percentile	Value
Group 1	34.4	17.1	0	12.7
			5	21.2
			10	26.8
			20	30.3
			25	31.4
			50	35.8
			75	38.9
			80	39.3
			90	40.5
			95	41.6
			100	43.6
Group 2	23.6	31.6	0	8.4
			5	11.9
			10	13.4
			20	17.8
			25	18.7
			50	23.5
			75	28.1
			80	30.0
			90	33.7
			95	36.0
			100	40.9

standard for embeddedness based on free matrix should be slightly higher than the measured criteria.

INTRAGRAVEL CONDITIONS IN SPAWNING AREAS

Sediment cores have been sampled in spawning areas in the SFSR, Secesh River, and Chamberlain Basin for sufficiently long to obtain a good picture of current conditions and trends in intragravel conditions. The SFSR and Secesh River sampling reflects conditions and trends in areas where there has been substantial resource development and some community development, whereas the Chamberlain Creek sites reflect wilderness conditions with minimal development and are used as reference sites. Because the reference sample is not very large and comes from a smaller set of watersheds and because multiple metrics from core samples can be measured, some differences in analytical approach have been used to take advantage of data collected from non-reference spawning areas; this will become clear as the analysis is explained.

Fine Sediments

Large Fines

Nonparametric analysis indicated significant differences among watersheds ($P < 0.0001$, 2 df), and the suggestion is that the SFSR sites likely average higher depth fines than the SECESH or CHAMB spawning areas (Figure 36). Our time series monitoring (Nelson *et al.* 1997 *et seq.*) has consistently shown the Lake Creek and Secesh River spawning areas (with the exception of one site near Threemile Creek) to be low in depth fines smaller than 6.3 mm diameter than the other two watersheds. There seems to be no compelling reason, however, to separate the watersheds to determine reference conditions, because the reference areas on Chamberlain Creek had depth fines approximately intermediate between the areas in the two developed watersheds. The relative positions of the medians for each distribution can be seen most clearly in Figure 37.

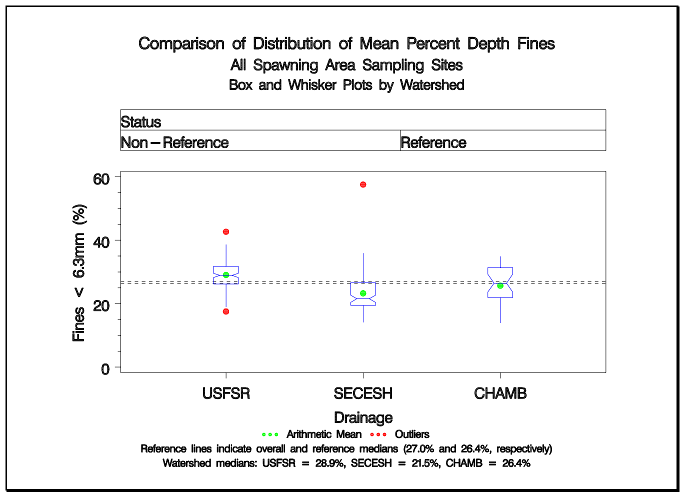

Figure 36.—Box and whisker plots comparing percent large fines distributions from spawning areas by watershed.

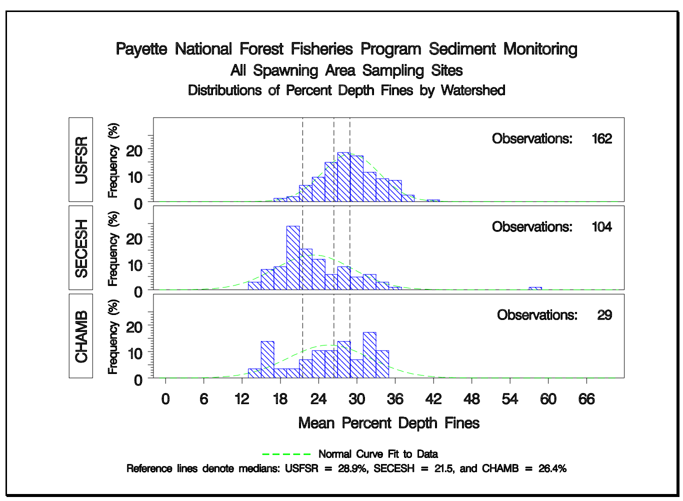

Figure 37.—Observed frequency distribution of spawning area depth fines means by watershed.

From the STE-large fines relationship shown in (Figure 18), the Secesh River sites would appear to have moderate STE potential for chinook salmon (~60%), the Chamberlain Basin sites would appear to have somewhat lower potential (~40%), and the upper SFSR would appear to have low potential (~10%).

Potentials would appear to be slightly greater using the relationship in Figure 19, but it should be expected that STE in an *in situ* experiment, given environmental stressors, variability in artificial redd packing, handling stress, and other uncontrollable factors, would be less than that observed in a laboratory experiment, even though green eggs may have been used in the latter while eyed eggs were used in the former.

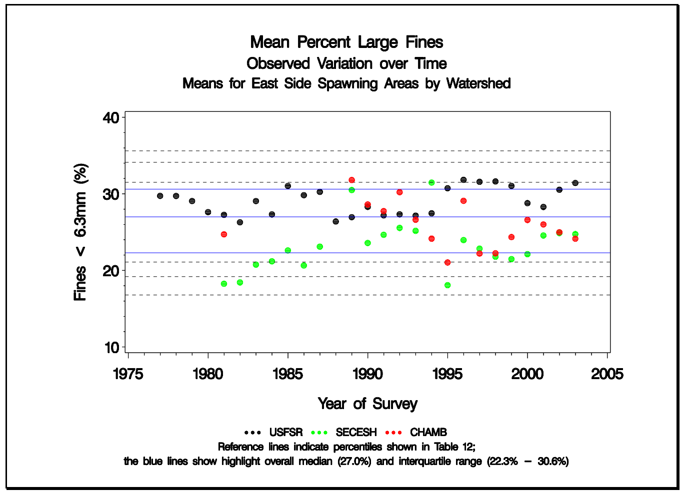

Figure 38.—Annual variation in mean depth fines (<6.3mm) for sampled watersheds relative to reference percentiles determined from the overall spawning area means.

Table 16.—Selected percentiles derived from the distribution of mean large fines (<6.3mm) in spawning areas (interquartile range is indicated by double lines).

Drainage Group	Mean	CV (%)	Per-centile	Value
Overall	26.6	22.2	0	13.9
			5	16.8
			10	19.2
			20	21.1
			25	22.3
			50	27.0
			75	30.6
			80	31.5
			90	34.1
			95	35.6
			100	57.5
Reference	25.6	25.1	0	13.9
			5	15.0
			10	15.3
			20	17.2
			25	21.9
			50	26.4
			75	31.4
			80	31.8
			90	33.4
			95	34.2
			100	34.9

The coefficients of variation for depth fines were somewhat smaller than seen with the other sediment measures (Table 16), but there was still substantial annual variation (Figure 38). A comparison of the means and percentiles for percent large fines overall and in the Chamberlain Basin is also presented in Table 16, which shows that the general similarity in the distributions resulting from the SFSR and SECESH sites averaging out in the overall analysis to provide a disturbed-area median similar to the reference condition. This can be seen in the comparison of the fluctuations (by watershed) over time, which also leads to the following generalizations:

- Conditions in the Chamberlain Basin are within the overall range (22.3% to 30.6%) about 88% of the time, but percent fines exceeded the 3rd quartile (75th percentile, 30.6%) once.

- Percent fines in the Secesh River sites are at or lower than the 1st quartile (25th percentile, 22.3%) about 43% of the time.

- At the SFSR sites, large fines were at or below the overall median (27.0%) only 3 times (12% of the time) and were at or above the 3rd quartile (30.6%) 8 times (31% of the time).

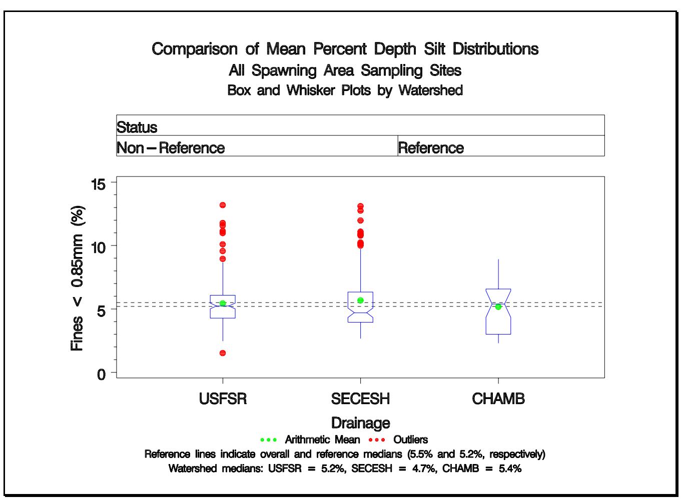

Figure 39.—Box and whisker plots comparing percent small fines distributions from spawning areas by watershed.

Small Fines

Percent of fine sediments smaller than 0.85 mm (small fines or silt) were not significantly different among watersheds (P = 0.5784, 2 df), and the similarities in the data distributions can be seen in Figures 39 and 40, former on previous page). The data (Table 17) suggest that about 5-6% small fines would represent the expected average natural condition for watersheds like these; the range seems to be greater in the developed watersheds, but that may be a reflection of the larger sample sizes. We have consistently reported a declining trend in small fine sediment concentration at the upper SFSR spawning areas we sample, and these were sometimes evident when large fine sediments seemed to be stable or even increasing somewhat (Nelson *et al.* 1997 *et seq.*). From the STE-small fines relationship shown in Figure 17, all spawning areas would appear to have low STE potential (~10%). This probably indicates that the curve in Figure 17 is shifted left relative to natural survival rates relative to our wet-sieved fines measurements (the same is probably true of the relationship in Figure 18 as well) because this survival rate would be quite low compared with values reported in *in situ* emergence studies (*e.g.*, Healey, 1991; Koski 1966; Meehan and Bjornn 1991; Sparkman 2003).

Table 17.—Selected percentiles derived from the distribution of mean silt (<0.85mm) in spawning areas (interquartile range is indicated by double lines).

Drainage Group	Mean	CV (%)	Per-centile	Value
Overall	5.5	37.9	0	1.5
			5	3.0
			10	3.5
			20	3.9
			25	4.1
			50	5.0
			75	6.2
			80	6.7
			90	8.5
			95	10.1
			100	13.2
Reference	5.2	40.2	0	2.3
			5	2.3
			10	2.3
			20	2.8
			25	3.0
			50	5.4
			75	6.6
			80	7.2
			90	8.3
			95	8.4
			100	8.9

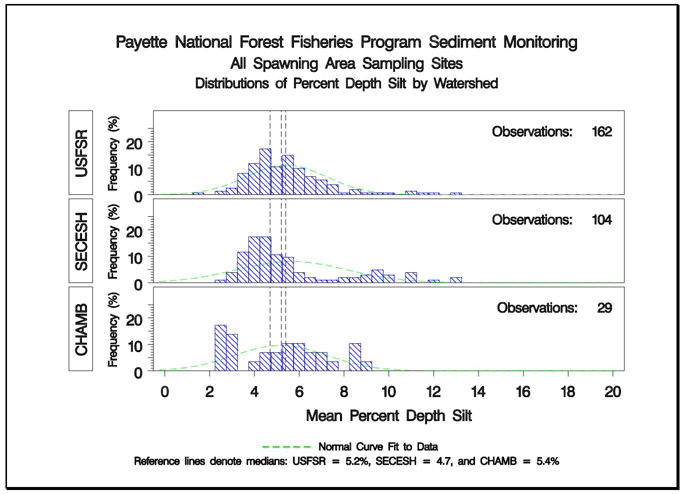

Figure 40.—Observed frequency distribution of spawning area depth silt means by watershed.

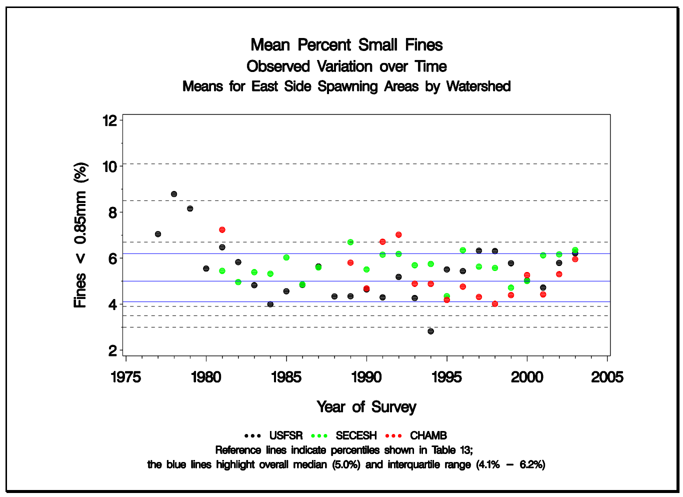

Figure 41.—Annual variation in mean depth fines (<0.85mm) for sampled watersheds relative to percentiles determined from the overall spawning area means.

Looking at the data distribution (Table 17) and the variation over time (Figure 41), we can make the following generalizations:

- Since 1980, the small fines levels in the SFSR have been within or below the overall interquartile range (4.1% to 6.2%) about 85% of the time.
- The SECESH sites have also been within the interquartile range at about the same frequency.
- Small fines in the reference watershed (CHAMB) have actually exceeded the 3rd quartile (6.2%) 19% of the time but was below the median (50th percentile, 5.0%) about 56% of the time.
- The SECESH areas seemed to have the least variability, while variability in CHAMB and USFSR were similar.

Intragravel Quality

Geometric Mean Particle Diameter

We have consistently reported geometric mean particle diameter (d_g) in addition to fine sediment concentrations to account for the controversy over what particle size classes are most important and to better describe the intragravel sediment composition. Geometric mean particle diameter was significantly different among watersheds ($P < 0.0001$, 2 df), apparently due to smaller median diameters in the upper SFSR spawning areas, though there was some small overlap in all the boxes (Figure 42). Despite having a lower concentration of large fines, on average, the Secesh sites had a slightly lower geometric mean diameter, on average, than the Chamberlain Basin sites, though we have not tested to determine whether the difference was statistically significant and the median for SECESH was somewhat larger. We attribute this situation to the relatively high concentration of small fines at the spawning site near Threemile Creek that is influenced by an unstable area that was mined around the turn of the century discussed previously and in other monitoring reports (Nelson *et al*. 1997 *et seq*.; a photograph of the disturbed area is included in Nelson *et al*. 2001). The distributional

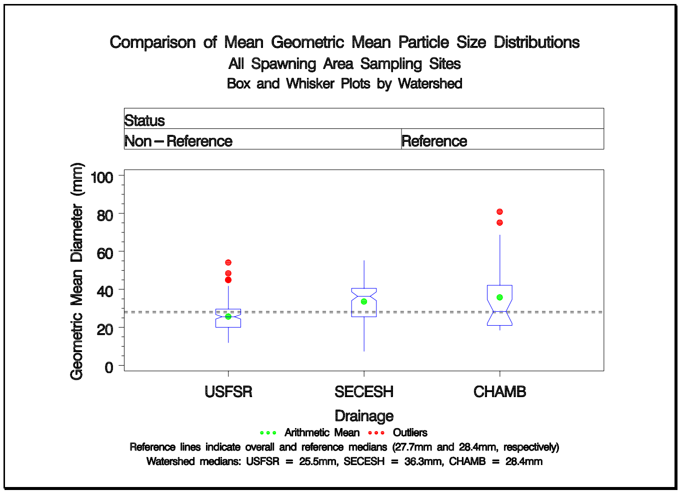

Figure 42.—Box and whisker plots comparing geometric mean particle diameter distributions from spawning areas by watershed.

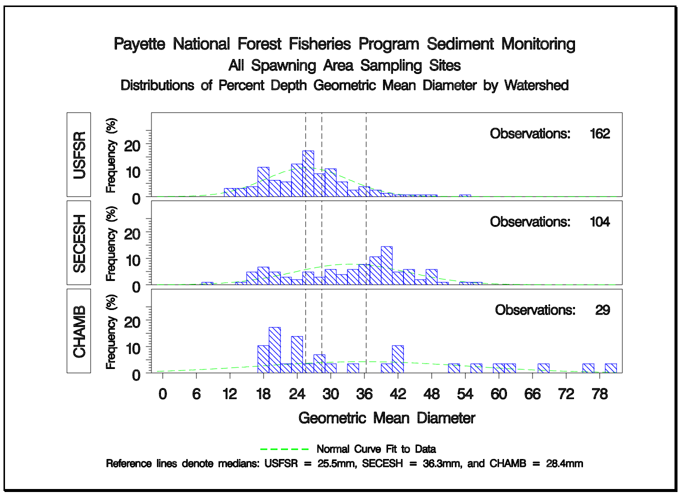

Figure 43.—Observed frequency distribution of spawning area geometric mean diameter means by watershed.

Table 18.—Selected percentiles derived from the distribution of geometric mean particle diameter in spawning areas (interquartile range is indicated by double lines).

Drainage Group	Mean	CV (%)	Per-centile	Value
Overall	29.5	36.8	0	7.3
			5	15.8
			10	17.7
			20	19.6
			25	22.0
			50	27.7
			75	36.5
			80	38.3
			90	42.5
			95	47.7
			100	80.9
Reference	35.8	52.4	0	18.4
			5	18.6
			10	18.8
			20	20.3
			25	20.9
			50	28.4
			75	42.2
			80	55.6
			90	68.8
			95	75.1
			100	80.9

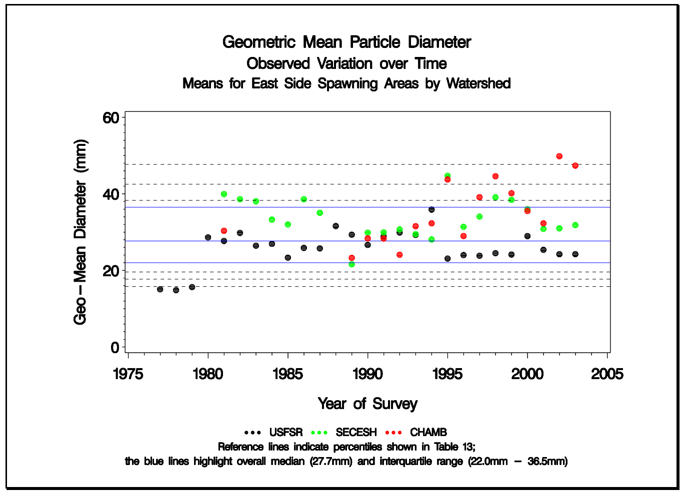

Figure 44.—Annual variation in mean geometric mean particle diameter for sampled watersheds relative to percentiles determined from the overall spawning area means.

similarities and differences are further displayed in Figure 43 (previous page), which shows the high variation in the CHAMB sites and the effects of a small sample size. We have elected to evaluate potential natural conditions from the overall distribution, which is similar to the reference distribution (Table 18) but provides a larger sample.

Looking at the geometric mean data distribution in Table 18 and the variation over time (Figure 44), we can make the following generalizations:

- The SFSR has not been below the lowest quartile (22.0mm) since 1980.
- The geometric mean particle diameter for the USFSR has been below the overall median value (27.7mm) about 64% of the time.
- The geometric mean diameter for CHAMB was never below the 1st overall quartile (22.0mm) and exceeded the 3rd quartile (36.5mm) about 35% of the time.
- The geometric mean diameter for SECESH was below the 1st reference quartile only once and was at or greater than the median about 95% of the time.
- CHAMB appeared to have the most annual variation, USFSR the least.

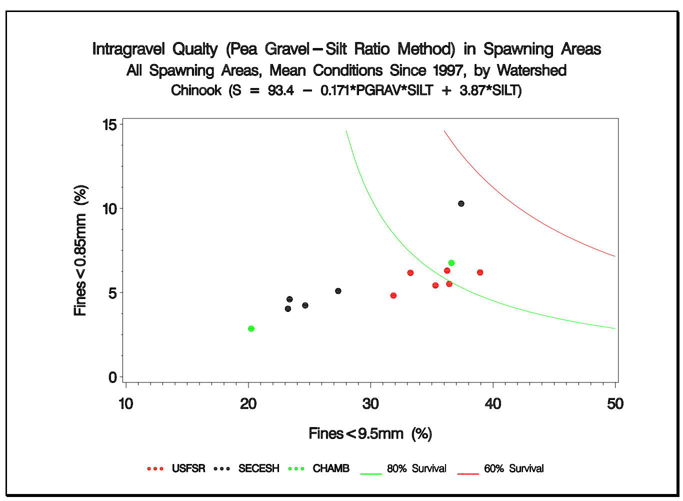

Figure 45.—Relationships of mean intragravel conditions to chinook embryo survival potential, 1997-2003 means.

Pea Gravel–Silt Ratio

The pea gravel–silt method, developed in 1983, appears to be a very nice way to illustrate intragravel quality, and we began using it in our sediment monitoring reporting in 1984 (see Lund [1984]); it has been a regular part of our comprehensive reports since 1996 (Nelson *et al*. 1996 *et seq*.). We have typically displayed the results of this analysis for each watershed and year of sampling, which is appropriate for annual monitoring reports. Here, however, it seemed more reasonable to display the overall averages since 1997, when the BNF added the required 9.5mm sieve, for this analysis.

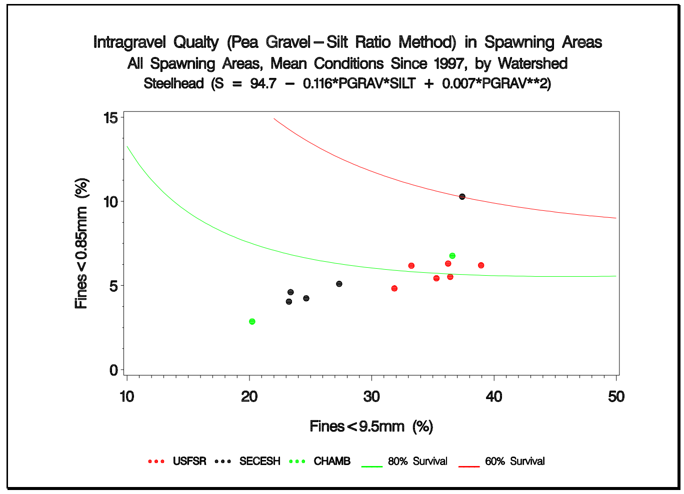

Figure 46.—Relationships of mean intragravel conditions to steelhead embryo survival potential, 1997-2003 means.

Clearly, most sites, on average, appear to have moderate to good survival potential for both Chinook (Figure 45, previous page) and steelhead (Figure 46) embryos. Quality seems to be somewhat higher relative to Chinook than for steelhead, despite the fact that steelhead are generally more tolerant of intragravel fines than Chinook. It should be noted that the 80% steelhead isoline, which is almost horizontal over much of the graph, and the right half of the 80% Chinook isoline, lies at about 6% small fines; this agrees approximately with our discussion above (Figure 17) suggesting that survival of embryos is likely to be highly compromised at concentrations of small fine sediments above 5 or 6%.

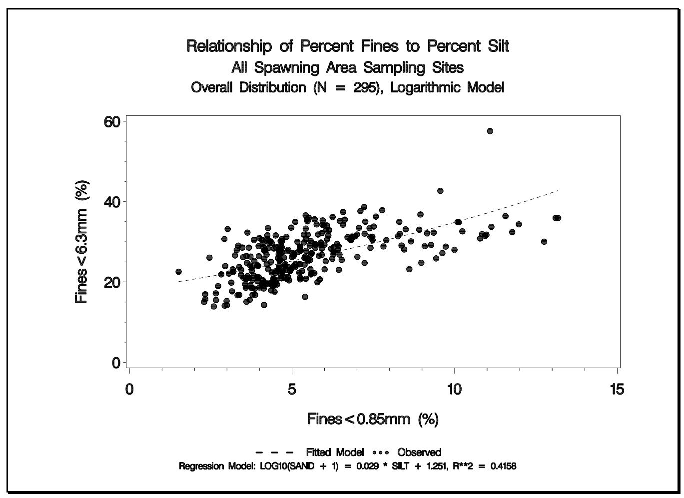

Figure 47.—Logarithmic regression of large fines on small fines from core samples.

Intragravel Quality Synthesis

The above discussions show that the upper SFSR spawning areas have more large fines than the reference sites in the Chamberlain Basin and the Lake Creek and Secesh River spawning areas, while also having essentially the same concentration of small fines. It is difficult to directly relate existing conditions to potential salmonid survival because it is not possible to directly extrapolate STE study results to field sediment monitoring efforts; this point is well made by Chapman and McLeod (1987). Different species of salmonids are also affected differently by fine sediments of different grain sizes (Irving and Bjornn 1984; Tappel and Bjornn 1983), suggesting that a "one size fits all" criterion is unrealistic. We can say conclusively, however, that the bull trout spawning and rearing sediment WCI of <12% intragravel sediments smaller than 0.85mm particle diameter proposed in the revised LRMP is inappropriate. In fact, regressing large fines on small fines from all samples (using site means) produced a significant log-linear relationship ($R^2 = 0.4158$, $P < 0.0001$)[27] that predicts about 38%

[27] The logarithmic model (using $\log_{10}$[fines + 1]) produced a slightly lower R^2 than the plain linear model (R^2 = 0.4215) but we selected it because it seems reasonable to expect large fines to increase faster than small

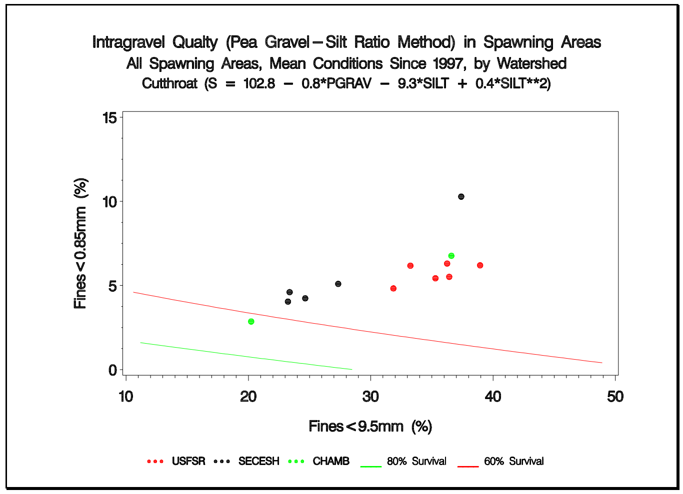

Figure 48.—Relationships of mean intragravel conditions to cutthroat trout embryo survival potential, 1997-2003 means.

large fines at 12% small fines (Figure 47, previous page). This is a very high concentration unlikely to be seen naturally even in low-gradient anadromous fish spawning areas. We have seen levels of large fines this high at times in some of the upper SFSR spawning areas, but we have not observed small fines levels over 10% since 1979 (Nelson *et al*. 1997 *et seq*.), which may have resulted, at least in part, from the exceptionally high runoff in the SFSR watershed in 1974[28]. Bull trout typically spawn in higher gradient tributary systems where fine sediment levels are likely to be lower than these traditional salmon and steelhead spawning areas. This can perhaps best be illustrated with the relationship for cutthroat trout, typically a tributary spawner, developed by Irving and Bjornn (1984) using our sediment data from these traditional spawning areas (Figure 48). This graph illustrates two key points:

- Favorable conditions for embryos of large anadromous species are not necessarily good for other resident fish that use tributaries for spawning.

- Resident fish may be adapted to streambed conditions characterized by even fewer fine sediments than salmon and steelhead use effectively.

fines because it reflects more size classes and because it seems to minimize the effect of the upper SFSR sites having a lower concentration of small fines relative to large fines than the other areas.

[28] The highest discharge ever recorded at the USGS gage near Krassel Guard station (6,740cfs) was recorded in the spring of 1974 (http://nwis.waterdata.usgs.gov/nwis/nwisman/?site_no=13310700&agency_cd=USGS).

REVISED SEDIMENT INDICES FOR WCIS

We established the distributions of the various sediment indices in order to see how they varied in time and space to facilitate suggesting revised values based on natural conditions and their quality with respect to salmonid production. Bauer and Ralph (2001) mention using quartiles (specifically the third quartile or 75th percentile) as a potential boundary for acceptable values of a generic water quality indicator, but they provide no additional information how to use these percentiles in establishing specific criteria. We have largely adopted this framework, although we have used it in conjunction with our knowledge of fish biology to set the actual index values. We have provided multiple mechanisms for determining sediment conditions and functional category that incorporate the opportunity to use few or several measurements; in all cases, a reduced sampling burden equates to a more stringent criterion. Median values from the distributions of sediment indices measured across time and space provide the basis for each multiple-sample index, with some exceedences in the first or third quartile, whichever corresponds to poorer conditions, allowed when multiple samples are used; single sample estimates, on the other hand, are restricted to the first or third quartile, whichever indicate more suitable habitat. This process has resulted in the likelihood that assessment of habitat conditions will lead to some FA calls in the SFSR, and implies that the SFSR may be inherently functioning at risk. We believe that this is appropriate given the fact that we know that sediment producing events that may depress salmonid production occur naturally and historic development did, in fact, interact with this innate character to catastrophically reduce potential productivity. We have attempted to avoid reliance on surface fines as a stand-alone indicator because we do not believe that it is a reliable estimator of streambed quality, but we have provided a preliminary index value for it in combination with free matrix that we suggest using on a trial basis. We have also rejected the use of geometric mean diameter as a basis for WCIs because of apparent discrepancies among methods of calculation and our inability to clearly relate measured values to potential salmonid embryo survival (discussed below). We believe that the graphical pea gravel-silt ratio is promising, but have not, at this time, attempted to apply it to the WCI functional categories.

These proposed WCIs were developed only for use in the granitic portions of the SFSR, although use in the Chamberlain Basin is also appropriate. We think it likely that they would be suitable for other granitic watersheds on the Forest as well, but applicability should be verified with local data. There will be situations where no local reference data can be obtained, in which case the data shown for non-reference conditions might provide a suitable benchmark. In addition, these indices should not be compared to sediment values obtained using other sampling protocols. We did not develop specific WCI values for the volcanic sections of the area analyzed, though it would be a simple matter to use the Group 2 distributions to determine what interstitial sediment values would be appropriate; we know of no core sampling data from those areas.

SURFACE FINES

Using a definition of surface fines as particles smaller than 6.3mm particle diameter, streams functioning appropriately would generally have surface fines levels of about 20% or less (the third quartile from reference data) in the SFSR, Secesh River, and Chamberlain Creek watersheds in the normally-encountered granitics, but not in different geologies as in Porphyry Creek and the EFSFSR. If we applied the same reasoning to consideration of what constitutes functioning at risk and functioning at unacceptable risk, ranges of 20% to 32% and greater than 32% would seem to be appropriate. The selected non-reference sites reviewed here were quite often in the FR range, whereas the reference sites rarely were; samples in FUR range would be uncommon, which seems to be a reasonable proposition. One sample would be insufficient, however, because our USFSR reference site had about a 30% chance of being in the FR range, so a five-year average would be more appropriate for accurately assessing condition. Because of the considerations discussed in Nelson *et al*. (2004b), however, we do not think that surface fines should be used as a stand-alone criterion. It would probably be reasonable, however, to combine surface fines measurements with an interstitial measurement (probably free matrix counts) such that fewer free matrix samples would be needed in order to reduce the risk that an inappropriate determination of condition would result; this is discussed in the section summary below.

COBBLE EMBEDDEDNESS

Using a definition of cobble embeddedness based on the embedding particles being particles smaller than 6.3mm in particle diameter, streams functioning appropriately would generally have an embeddedness value of about 32% in the SFSR, Secesh River, and Chamberlain Creek watersheds in the normally-encountered granitics, but not in different geologies as in Porphyry Creek and the EFSFSR. If we applied the same reasoning to consideration of what constitutes functioning at risk and functioning at unacceptable risk, ranges of 32% to 42% and greater than 42%, with occasional exceedences, would seem to be appropriate. The selected non-reference sites reviewed here were quite often in the FR range, whereas the reference sites rarely were; samples in FUR range would be uncommon in undeveloped sites, which seems to be a reasonable proposition. One sample would be less than desirable, however, because our USFSR reference site had about a 50% chance of being in the FR range, so a five-year average would be more appropriate for accurately assessing condition. We suggest that a single sample would call for lower indices (*i.e.*, the first quartile) for FA, to minimize risk with the small sampling effort.

The preceding interpretation is similar, to that proposed in Nelson *et al*. (1997), despite being based on a differently designed analysis and a larger data set. Because of problems we have identified with cobble embeddedness measurements, it may be better to eschew cobble embeddedness in favor of an estimated value predicted from free matrix counts or free matrix counts alone, using a smaller, carefully measured cobble embeddedness sampling for regression and quality control purposes. In this case, the embeddedness indicator values would be different because of differences in the distributions of measured embeddedness and free matrix; we have included these along with the potential surface fines indicator as criteria for further evaluation in the section summary below.

FREE MATRIX COUNTS

Sampling in reference sites suggests that free particles will comprise about 11% or more of the sample most of the time, but may be less than this about 30% of the time in the SFSR, Secesh River, and Chamberlain Creek watersheds in the normally-encountered granitics, but not in different geologies as in Porphyry Creek and the EFSFSR. If we applied the same reasoning to consideration of what constitutes functioning at risk and functioning at unacceptable risk, ranges of 5% to 11% and less than 5% would seem to be appropriate. This would require a multi-year average, however, because even the reference site Blackmare Creek was below 5% about 13% of the time; the non-reference site on Buckhorn Creek was below 5% at about the same frequency, but averaged slightly fewer free particles. This is more generous than proposed in Nelson *et al*. (1997) but is similar and based on a more thorough analysis.

INTRAGRAVEL QUALITY

Core sampling is probably the best method for determining the ability of streambeds in primary Chinook and steelhead spawning areas to provide suitable conditions for spawning and embryo development. These areas also provide a location that integrates the effects of disturbances distributed throughout relatively large watershed areas. The LRMP uses depth fines as the criteria for functional classes of bull trout spawning and rearing and does not use them for Chinook and steelhead. Core sampling is likely inappropriate as a general sampling method in bull trout spawning areas because the technique is best suited to larger streams with distinct spawning areas to generate the 30-40 samples needed per year, and works well for Chinook and steelhead spawning and incubation habitat.

Our analysis suggests that large fine sediments up to about 31% (approximately the 3rd reference quartile) are likely to occur with regularity. One problem with our control watershed (CHAMB) is that it is not clearly similar to what the upper SFSR might have been like prior to development, and it is probably less subject to the hillslope failures that routinely occur in the SFSR; the Secesh River watershed may have been more similar, but most of the sampled spawning areas are also located in places less subject to mass failures. Stowell *et al*. (1983) present a model for the SFSR spawning areas (Figure 49) that suggests a pre-disturbance large fines concentration of about 25%, which comports well with the overall median from our analysis. If we further assume that the SFSR is currently functioning at risk for Chinook and steelhead embryo survival with respect to large fine sediments, an assumption that seems reasonable given the intragravel quality relationships shown in Figures 45 and 46, we are led to the conclusion that FA agrees approximately with 5-year mean large fines concentrations at or below 28% and no more than two single years greater than 36%, FR with a 5-year mean range of 28% to 36% with no more than two years over 36%, and FUR is anything higher.

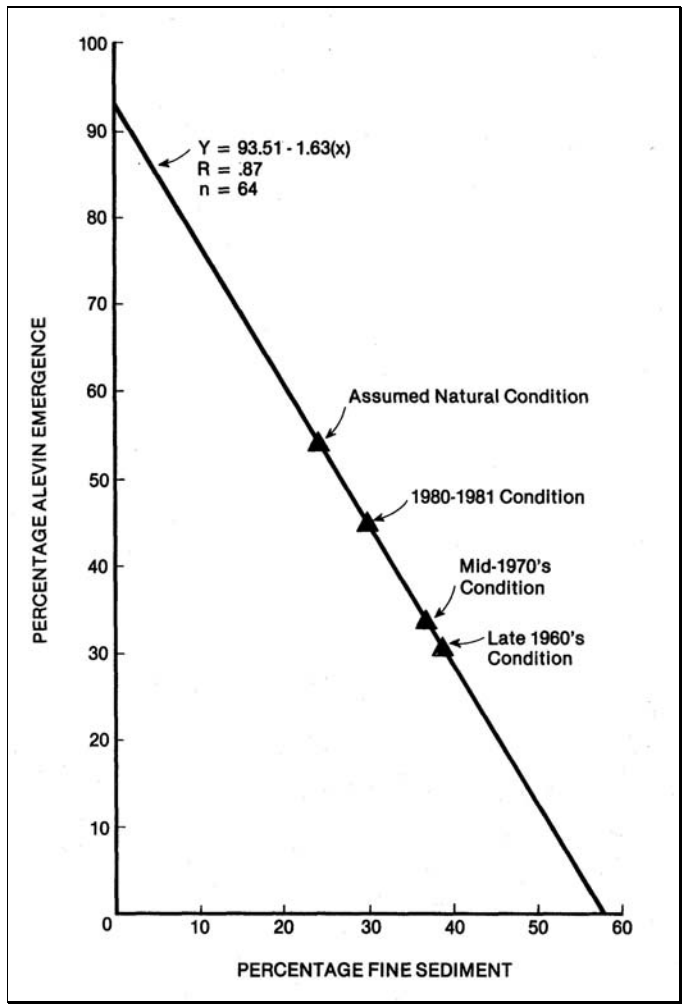

Figure 49.—Graph showing estimated pre-1965 condition of the SFSR with respect to large fine sediments (from Stowell *et al*. 1983).

We would not recommend using WCIs based on small fine sediments because the range where survival is optimized is so narrow and sampling errors could have large effects on the results. Geometric mean diameter would be a suitable alternative, but is probably of no more value in this use than large fines, and may be less so because it is less commonly seen in the literature and there is some question as to whether our calculations are consistent with published values. For example, Shirazi and Seim (1979) provide an STE curve for salmonid embryos at varying levels of d_g (Figure 50) that suggests nearly optimal survival of steelhead embryos at our median values for d_g; this doesn't seem realistic and may indicate that our calculations of d_g are either flawed or calculated differently from those that contributed to this figure.

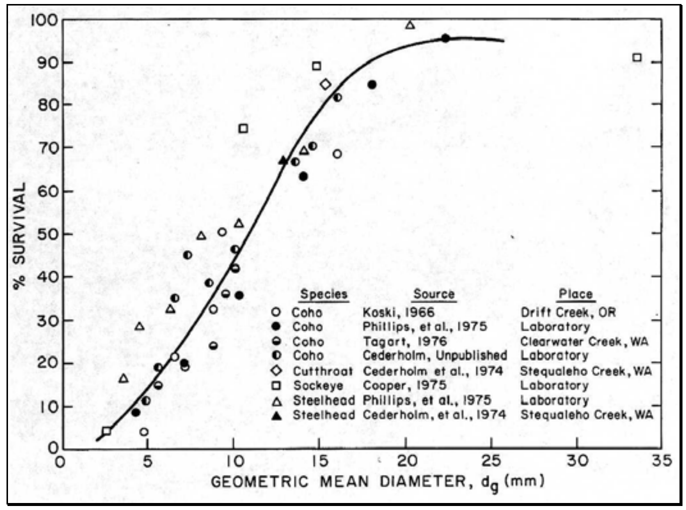

Figure 50.—Relationships of salmonid STE as a function of geometric mean particle diameter (from Shirazi and Seim 1979).

Summary

It is important to note that the WCIs below can only be used where there is a known relationship between the type of disturbance being evaluated and fine sediment deposition in fish habitat. It is inappropriate to assume that predicted changes in sediment yields due to fire or sheep grazing will result in any change in fine sediment deposition in fish habitat. It may be appropriate to infer that changes can occur due to management activities like road construction, or reconstruction, mining, and similar mechanical disturbance. We have pointed that out in the introduction and reiterate here.

Based on the preceding analysis using local data from a large number of sites in reference areas with some comparison to non-reference situations, the revised WCIs shown in Table 19 (below) seem reasonable. It may be incomplete, in that the "Substrate Embeddedness" WCI was omitted as an independent WCI; we do not see the need for multiple WCIs based on sediment conditions. This may seem to be a substantial change from the LRMP because bull trout are no longer identified in a WCI separate from anadromous species. In application, however, we do not identify bull trout rearing areas, which are also used by anadromous juveniles, so the WCI in the matrix would be applied to tributary systems generally; this has simply been codified in Table 20. Additional, preliminary definitions of the interstitial sediment indicators are presented in Table 20 (below) for use on a trial (*i.e.*, verification) basis. We envision using Table 20 in concert with Table 19 to see whether they result in the same determinations of functional condition for some period and revising as necessary. Although this table, if determined to be adequate, eliminates the possibility of relying on surface fines alone for determination of condition, not as a stand-alone indicator, it allows for reduced sampling relative to 5-year indices; it may also be a useful table for clarification of condition when small-sampling indicators from Table 16 are used.

We recognize that the proposed criteria might appear to make higher sediment levels acceptable than usually supposed; we believe that this is at least partly a consequence of attempting to work with artificial categories that may not adequately model the real world. In other words, we have had to place values of continuous variables in somewhat arbitrary categories. We have tried to accommodate this conundrum by allowing evaluation of conditions at multiple scales, the simplest being a single sampling that has more risk of incorrectly interpreting actual conditions, to multiple sampling in space or time that is more likely to estimate true conditions; the former values are more conservative than the latter.

True differences between reference and non-reference conditions can be masked somewhat by natural variation as well, which might lead to the appearance that watersheds with clear management-related disturbances have average conditions that are similar to reference conditions. While this may be true in some cases, differences can become evident when looking at temporal variation in conditions. This has been conceptualized by the relationship diagrammed in Figure 51. This sort of relationship was shown by our analysis in the comparisons of cobble embeddedness and free matrix counts at Blackmare Creek (reference) and Buckhorn Creek (managed with extensive rehabilitation work), where temporal fluctuations appeared to be higher in the former but the average index values were similar. Dealing with this situation is outside the scope of this paper, but it underscores the problem with assigning continuous, variable attributes to arbitrary functional categories. Our assignments of WCI values and ranges seem more reasonable than those promulgated by the revised LRMP, but there remains some disconnection with the reality of ecological processes, which have implicitly been treated as hierarchical but are more likely chaotic.

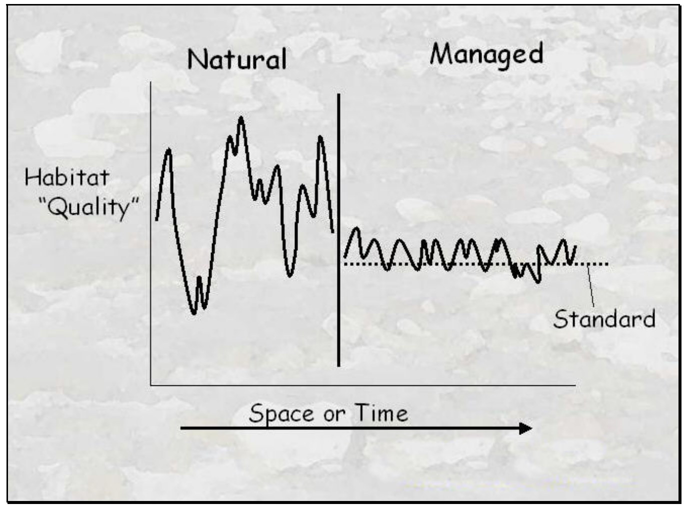

Figure 51.—Conceptual differences in variation in habitat conditions between managed and natural environments (from workshop presentation associated with Rieman *et al.* [2002]).

We have performed this analysis specifically to correct WCIs for use in the SFSR, which underlain is predominantly by granitic rocks. However, the data used were drawn from the Chamberlain Basin and parts of the Edwardsburg area as well, both of which are also predominantly granitic watersheds. It is important to avoid extrapolating the results of one study outside the range of the data analyzed, but it seems reasonable to suppose that the results of this effort would minimally apply to the Chamberlain Basin and upper Big Creek areas as well as to the SFSR, and may be more appropriate in granitic areas generally than the default WCIs in the revised LRMP.

Table 19.—Proposed revised watershed condition indicators (WCIs) for table B-1 of the revised PNF LRMP, expressed as functions of surface or depth fines, cobble embeddedness, and free matrix counts. Use of this table assumes the following: (1) all data QA/QC procedures from sediment reports were used; (2) the CE-FM regression indicates valid data were collected; (3) where multiple metrics are available, the ones indicating the highest sediment levels are used; (4) the longest time interval available is used with the most recent data; (5) data from the nearest downstream sites are used; and (6) analysis not clearly discriminating between two functional classes indicates that the lower class be used.

Pathways and WCIs	***Functioning Appropriately***	***Functioning at Risk***	***Functioning at Unacceptable Risk***
Interstitial Sediment Deposition (all listed fishes in tributary systems)	Adequate interstitial space is indicated by: (a) Any single **measured** mean embeddedness value less than or equal to 24%. **OR** (b) Any single mean free matrix count over 27% **OR** (c) A five-year mean **measured** cobble embeddedness level of 32% or less **OR** (d) A five-year mean free matrix count of 17% or more.	Reduced interstitial space is indicated by: (a) Any single **measured** mean embeddedness value between 24% and 32%. **OR** (b) Any single mean free matrix count between 17% and 27% **OR** (c) A five-year mean **measured** cobble embeddedness level of 32% to 42% **OR** (d) A five-year mean free matrix count of 11% to 17%.	Inadequate interstitial space is indicated by: (a) Any single **measured** mean embeddedness value over 32%. **OR** (b) Any single mean free matrix count less than 17% **OR** (c) A five-year mean **measured** cobble embeddedness level greater than 42% **OR** (d) A five-year mean free matrix count of less than 11%.
Interstitial Sediment Deposition (other fish species: *i.e.*, red band, rainbow, wood river sculpin, *etc.*)	For salmonids, use same as for listed species, develop criteria for other species as needed.		
Intragravel Quality (in areas of spawning and incubation for anadromous fishes)	High intragravel quality is indicated by: (a) 5-year mean fines < 6.3 mm concentrations at depth of 28% or less with no more than two years between 28% and 36%. **OR** (b) 5-year mean fines < 6.3 mm concentrations at depth between 28% and 36% with a **decreasing** trend over at least 10 years.	Moderate intragravel quality is indicated by: (a) 5-year mean fines < 6.3 mm concentrations at depth 28% to 36% with no more than two years > 36%. **OR** (b) 5-year mean fines < 6.3 mm concentrations at depth between 28% and 36% with an **increasing** trend over at least 10 years. **OR** (c) 5-year mean fines < 6.3 mm concentrations at depth of 36% or more with a **decreasing** trend over at least 10 years.	Low intragravel quality is indicated by: (a) 5-year mean fines < 6.3 mm concentrations at depth of 36% or more. **OR** (b) 5-year mean fines < 6.3 mm concentrations at depth 36% or more with an **increasing** trend over at least 10 years.
Substrate Embeddedness (Bull trout rearing areas. Spawning and incubation areas are addressed under the Sediment/Turbidity WCI)	Replaced with Interstitial Sediment Deposition for Listed Fishes WCI above		

Table 20.—Proposed revised watershed condition indicators (WCIs) for table B-1 of the revised PNF LRMP, expressed as functions of surface or depth fines, cobble embeddedness, and free matrix counts. Use of this table assumes the following: (1) all data QA/QC procedures from sediment reports were used; (2) the CE-FM regression indicates valid data were collected; (3) where multiple metrics are available, the ones indicating the highest sediment levels are used; (4) the longest time interval available is used with the most recent data; (5) data from the nearest downstream sites are used; and (6) analysis not clearly discriminating between two functional classes indicates that the lower class be used.

Pathways and WCIs	***Functioning Appropriately***	***Functioning at Risk***	***Functioning at Unacceptable Risk***
Interstitial Sediment Deposition (all listed fishes in tributary systems)	Pending verification, adequate interstitial space may also indicated by: (e) A two-year mean surface fines level of 20% or less **AND** A two-year mean measured cobble embeddedness level of 32% or less.	Pending verification, reduced interstitial space may also indicated by: (e) A two-year mean surface fines level of 20% to 32% **AND** A two-year mean measured cobble embeddedness level of 42% to 55%.	Pending verification, inadequate interstitial space is also indicated by: (e) Any combination of surface fines and cobble embeddedness measurements that indicate higher interstitial sediment than FR.
	OR (f) A two-year mean surface fines level of 20% or less **AND** A two-year mean free matrix count of 17% or more.	**OR** (f) A two-year mean surface fines level of 20% to 32% **AND** A two-year mean free matrix count of 11% to 27%	**Examples:** Surface Fines = 35% and Free Matrix = 10% Surface Fines = 35% and Embeddedness = 40%
	OR (g) Any single **predicted** mean embeddedness value less than or equal to 31%.	**OR** (g) Any single **predicted** mean embeddedness value between 31% and 36%.	**OR** (g) Any single **predicted** mean embeddedness greater than 36%.
	OR (h) A five-year mean predicted cobble embeddedness level of 36% or less	**OR** (h) A five-year mean cobble embeddedness level of 36% to 40%	**OR** (h) A five-year mean cobble embeddedness level greater than 40%

References

NOTE: Internet (WWW) and intranet (FSWeb) links to on-line references are provided for convenience. The links worked properly when this report was prepared, but the authors make no guarantee that they will work after this report is released. Obviously, the intranet links are only available to qualified persons.

Bauer, S.B.; Ralph, S.C. 2001. Strengthening the use of aquatic habitat indicators in Clean Water Act programs. Fisheries. 26(6):14-25.

Bjornn, T.C.; Brusven, M.A.; Molnau, M.P.; Milligan, J.H.; Klamt, R.; Chacho, E.; Schaye, C. 1977. Transport of granitic sediment in streams and its effects on insects and fish. Research Technical Completion Report, Project B-036-IDA. Submitted to U.S. Department of the Interior, Office of Water Research and Technology, Washington, DC. Moscow, ID: University of Idaho, College of Forestry, Wildlife and Range Sciences. 43p.

Bowman, K.; Minshall, G.W. 1999. Assessment of biota and habitat in tributaries of Big Creek and the South Fork Salmon River, Payette National Forest. Pocatello, ID: Idaho State University, Department of Biological Sciences, Stream Ecology Center. 42p.

Bowman, K.; Royer, T.; Minshall; G.W. 1998. Monitoring the range of ecological variability in burned and unburned streams of the Frank Church 'River of No Return' Wilderness during 1997. Pocatello, ID: Idaho State University, Department of Biological Sciences, Stream Ecology Center. 30p.

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. General Technical Report RMRS-GTR-74. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 428p.

Burns, D.C. 1987. Embeddedness of salmonid habitat of selected streams on the Payette National Forest 1985-1986. Unpublished report. McCall, ID: U.S. Department of Agriculture, Forest Service, Payette National Forest. 20p.

FSWEB:
http://fsweb.payette.r4.fs.fed.us/units/fish.web/sediment_reports/PNF_Interstitial_Sampling_Report_1987.pdf

Burns, D.C. 1984. An inventory of embeddeness of salmonid habitat in the South Fork Salmon River drainage, Idaho. Unpublished report. McCall, ID: U.S. Department of Agriculture, Forest Service, Payette National Forest. 30p.

FSWEB:
http://fsweb.payette.r4.fs.fed.us/units/fish.web/sediment_reports/PNF_Interstitial_Sampling_Report_1984.pdf

Burns, D.C.; Edwards, R.E. 1985. Embeddeness of salmonid habitat of selected streams on the Payette national Forest. Unpublished report. McCall, ID: U.S. Department of Agriculture, Forest Service, Payette National Forest. 40p.

FSWEB:
http://fsweb.payette.r4.fs.fed.us/units/fish.web/sediment_reports/PNF_Interstitial_Sampling_Report_1985.pdf

Burton, T.A.; Harvey, G.W.; McHenry, M.L. 1990. Protocols for assessment of dissolved oxygen, fine sediment and salmonid embryo survival in an artificial redd. Water quality monitoring protocols - report no. 1. Unpublished report. Boise, ID: Department of Health and Welfare, Division of Environmental Quality, Water Quality Bureau. 25p.

Burton, T.A.; Cowley, E.; Harvey, G.W.; Wicherski, B. 1991. Protocols for evaluation and monitoring of stream/riparian habitats associated with aquatic communities in rangeland streams. Water quality monitoring protocols - report no. 4. Draft. Boise, ID: Department of Health and Welfare, Division of Environmental Quality, Water Quality Bureau. Unnumbered pages.

Chamberlin, T.W.; Harr, R.D.; Everest, F.H. 1991. Timber harvesting, silviculture, and watershed processes. Chapter 6 in: Meehan, W.R. 1991. Influences of forest and rangeland management on salmonid fishes and their habitat. American Fisheries Society Special Publication 19: 181-205. Bethesda, MD: American Fisheries Society.

Chapman, D.W.; McLeod, K.P. 1987. Development of criteria for fine sediment in the Northern Rockies ecoregion. EPA 910/9-87-162. Seattle, WA: U.S. Environmental Protection Agency. 279p.

Cline, R.; Cole, G.; Megahan, W.; Patten, R.; Potyondy, J. 1981. Guide for predicting sediment yields from forested watersheds. Missoula, MT, and Ogden, UT: U.S. Department of Agriculture, Forest Service, Northern and Intermountain Regions. Pages variously numbered.

Corley, D.R. 1976. Fishery habitat survey of the South Fork Salmon River – 1975. Unpublished report. Boise, ID: U.S. Department of Agriculture, Forest Service, Boise National Forest. 70p.

Crouse, M.R.; Callahan, C.A.; Malueg, K.W.; Dominguez, S.E. 1981. Effects of fine sediments on growth of juvenile coho salmon in laboratory streams. Transactions of the American Fisheries Society. 110:281-286.

Edwards, R.E.; Burns, D.C. 1986. Relationships among fish habitat embeddedness, geomorphology, land disturbing activities and the Payette National Forest sediment model. Unpublished report. McCall, ID: U.S. Department of Agriculture, Forest Service, Payette National Forest. 12p.

FSWEB:
http://fsweb.payette.r4.fs.fed.us/units/fish.web/sediment_reports/PNF_Interstitial_Sampling_Report_1986.pdf

Furniss, M.J.; Roelofs, T.D.; Yee, C.S. 1991. Road construction and maintenance. Chapter 8 in: Meehan, W.R. 1991. Influences of forest and rangeland management on salmonid fishes and their habitat. American Fisheries Society Special Publication 19: 297-323. Bethesda, MD: American Fisheries Society.

Healey, M.C. 1991. Life history of Chinook salmon (*Oncorhynchus tshawytscha*). In: Groot, C.; Margolis, L. (ed). Pacific salmon life histories. Vancouver, BC: University of British Columbia Press. p313-393.

Heikkenen, E. No Date. An overview of events that transpired on the South Fork Salmon River--Particular reference to the anadromous fish values as they were affected by Forest Service land management activities. Unpublished original transcription of report by the District Ranger at Krassel on file. McCall, ID: U.S. Department of Agriculture, Forest Service, Payette National Forest. 16p.

Hockaday, J.M. 1968. History. Payette National Forest. Unpublished report. McCall, ID: U.S. Department of Agriculture, Forest Service, Payette National Forest. 175p.

Hurlbert, S.H. 1984. Pseudoreplication and the design of ecological field experiments. Ecological Monographs. 54(2):187-211.

Irving, J.S.; Bjornn, T.C. 1984. Effects of substrate size composition on survival of kokanee salmon and cutthroat and rainbow trout embryos. Completion report. Cooperative Agreement No. 12-11-204-11, Supplement No. 87. Presented to U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station, Boise, Idaho. Moscow, ID: University of Idaho, College of Forestry, Wildlife and Range Sciences. 21p.

Jensen, F.; Cole, G.F. 1965. The South Fork of the Salmon River storm report. Unpublished report. McCall, ID: U.S. Department of Agriculture, Forest Service, Payette National Forest. 15p. (plus attachments).

FSWEB: http://fsweb.payette.r4.fs.fed.us/units/fish.web/flood_reports/sfsr_flood_report_65.pdf

Kelley, D.W.; Dettman, D.H. 1980. Relationships between streamflow, rearing habitat, substrate conditions, and juvenile steelhead populations in Lagunitas Creek, Marin County, 1979. Unpublished report to Marin Municipal Water District. Sacramento, CA: D.W. Kelley, Aquatic Biologist.

Kershner, J.L.; Coles-Ritchie, M.; Cowley, E.; Henderson, R.C.; Kratz, K.; Quimby, C.M.; Turner, D.M.; Ulmer, L.C.; Vinson, M.R. no date. A plan to monitor the aquatic and riparian resources in the area of PACFISH/INFISH and the biological opinions for bull trout, salmon, and steelhead. Unpublished report. Logan, UT: U.S. Department of Agriculture, Forest Service, Fish and Aquatic Ecology Unit. 103p.

WWW: http://www.fs.fed.us/biology/resources/pubs/feu/pibo_final_011003.pdf

Kershner, J.L.; Coles-Ritchie, M.; Cowley, E.R.; Henderson, R.C.; Kratz, K.; Quimby, C.; Turner, D.L.; Ulmer, L.C.; Vinson, M.R. 2004a. Guide to effective monitoring of aquatic and riparian resources. General Technical Report RMRS-GTR-121. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 57p.

WWW: http://www.fs.fed.us/biology/resources/pubs/feu/rmrs_gtr_121.pdf

Kershner, J.L.; Roper, B.B.; Bouwes, N.; Henderson, R.; Archer, E. 2004b. An analysis of stream habitat conditions in reference and managed watersheds on some federal lands within the Columbia River basin. North American Journal of Fisheries Management. 24:1363-1375.

WWW: http://www.fs.fed.us/biology/resources/pubs/feu/04refmanagedfinalNAJFM.pdf

Klamt, R.R. 1976. The effects of coarse granitic sediment on the distribution and abundance of salmonids in the central Idaho Batholith. MS Thesis. Moscow, ID: University of Idaho, College of Forestry, Wildlife and Range Sciences. 84p.

Koski, K.V. 1966. The survival of coho salmon (*Oncorhynchus kisutch*) from egg deposition to emergence in three Oregon coastal streams. MS thesis. Corvallis, OR: Oregon State University. 84p.

Lund, J. 1982. Core sample analyses of fine sediment in spawning sites, Lake Creek–Secesh River – 1981. Unpublished report. McCall, ID: U.S. Department of Agriculture, Forest Service, Payette National Forest. 16p.

FSWEB: Not yet posted.

McNeil, W.J. 1964. A method of measuring mortality of pink salmon eggs and larvae. Fisheries Bulletin 63:575-578. Washington, DC: U.S. Department of the Interior, Fish and Wildlife Service.

Meehan, W.R.; Bjornn, T.C. 1991. Salmonid distributions and life histories. In: Meehan, W.R. (ed). Influences of frest and rangeland management on salmonid fishes and their habitat. Bethesda, MD: American Fisheries Society. p47-82.

Minshall, G.W.; Robinson, C.; Royer, T. 1994. Biomonitoring results from wilderness streams in Idaho. Pocatello, ID: Idaho State University, Department of Biological Sciences, Stream Ecology Center. 32p.

Nelson, R.L.; Burns, D.C.; Ketchu, K.L. 2004a. Relationships between estimated clearcut area and fish habitat attributes in selected streams on the Payette National Forest, with application to use of watershed condition indicators in forest planning and consultation under the Endangered Species Act. Unpublished white paper. McCall, ID: U.S. Department of Agriculture, Forest Service, Payette National Forest. 63p.

FSWEB: http://fsweb.payette.r4.fs.fed.us/units/fish.web/white_papers/ECA_Fish-Habitat.pdf

Nelson, R.L.; Burns, D.C.; Newberry, D.D.; Ketchu, K.L. 2004c. Deposition of fine sediment in selected streams on the Payette and Boise National Forests, Idaho. Intragravel conditions in spawning areas. Statistical summary of intragravel monitoring, 1975-2003, and photographic summary of south fork salmon river changes since 1966. McCall, ID: U.S. Department of Agriculture, Forest Service, Payette National Forest. 173p.

FSWEB:
http://fsweb.payette.r4.fs.fed.us/units/fish.web/sediment_reports/PNF_Core_Sampling_Report_2004.pdf

Nelson, R.L.; Burns, D.C. 2004d. Deposition of fine sediment in selected streams on the Payette and Boise National Forests, Idaho. Interstitial conditions in Salmon River tributaries. Statistical summary of interstitial and surface sediment monitoring, 1983-2003. Unpublished report. McCall, ID: U.S. Department of Agriculture, Forest Service, Payette National Forest. 331p (plus errata).

FSWEB:
http://fsweb.payette.r4.fs.fed.us/units/fish.web/sediment_reports/PNF_Interstitial_Sampling_Report_2003.pdf

Nelson, R.L.; Burns, D.C.; Ketchu, K.L. 2003a. Preliminary analysis evaluating potential relationships of equivalent clearcut area to fish habitat attributes in selected streams on the Payette National Forest. Draft white paper. McCall, ID: U.S. Department of Agriculture, Forest Service, Payette National Forest. 39p.

Nelson, R.L.; Burns, D.C.; Ketchu, K.L. 2003b. Deposition of fine sediment in selected streams on the Payette and Boise National Forests, Idaho. Interstitial conditions in Salmon River tributaries. Report of sediment trends from cobble embeddedness, free matrix, and surface fines, 1981-2001. Unpublished report. McCall, ID: U.S. Department of Agriculture, Forest Service, Payette National Forest. 99p.

FSWEB:
http://fsweb.payette.r4.fs.fed.us/units/fish.web/sediment_reports/PNF_Interstitial_Sampling_Report_2003.pdf

Nelson, R.L.; Burns, D.C.; Ketchu, K.L.; Newberry, D.D. 2002. Deposition of fine sediment in selected streams on the Payette and Boise National Forests, Idaho. Intragravel conditions in spawning areas. Report of sediment trends from core sampling, 1966-2001. Unpublished report. McCall, ID: U.S. Department of Agriculture, Forest Service, Payette National Forest. 69p.

FSWEB:
http://fsweb.payette.r4.fs.fed.us/units/fish.web/sediment_reports/PNF_Core_Sampling_Report_2002.pdf

Errata Sheet:

FSWEB:
http://fsweb.payette.r4.fs.fed.us/units/fish.web/sediment_reports/pnf_core_sampling_report_2002_errata.pdf

Nelson, R.L.; Burns, D.C.; Ketchu, K.L.; Newberry, D.D. 2001. Deposition of fine sediment in selected streams on the Payette and Boise National Forests, Idaho. Intragravel conditions in spawning areas. Report of sediment trends from core sampling, 1966-2000. Unpublished report. McCall, ID: U.S. Department of Agriculture, Forest Service, Payette National Forest. 76p.

FSWEB:
http://fsweb.payette.r4.fs.fed.us/units/fish.web/sediment_reports/PNF_Core_Sampling_Report_2001.pdf

Errata Sheet:

FSWEB:
http://fsweb.payette.r4.fs.fed.us/units/fish.web/sediment_reports/PNF_Core_Sampling_Report_2001_errata.pdf

Nelson, R.L.; Burns, D.C.; Newberry, D.D. 1998. Spawning area sediment trends in the South Fork Salmon River. Unpublished report. McCall, ID: U.S. Department of Agriculture, Forest Service, Payette National Forest. 30p.

Nelson, R.L.; McGee, M.N.; Zurstadt, C.F.; Burns, D.C. 2004b. Evaluation of surface fines as an index of salmonid habitat conditions on the Payette National Forest.

Unpublished white paper. McCall, ID: U.S. Department of Agriculture, Forest Service, Payette National Forest. 17p.

FSWEB: http://fsweb.payette.r4.fs.fed.us/units/fish.web/white_papers/surface_fines_report.pdf

Nelson, R.L.; Platts, W.S. No Date. Testing and refining the Fisheries–Sediment Response model. Preliminary findings. Unpublished report. Boise, ID: U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station. 27p.

Nelson, R.L.; Wagoner, L.J.; Burns, D.C.; Newberry, D.D.; Lund, J. 1997. Deposition of fine sediment in selected streams on the Payette and Boise National Forests, Idaho. Report of sediment trends and monitoring efforts, 1977-1996. Trends in cobble embeddedness, free matrix particles, surface fines, and intragravel conditions in the Chamberlain Basin and the South Fork Salmon River and selected tributaries. Unpublished report. McCall, ID: U.S. Department of Agriculture, Forest Service, Payette National Forest. 199p.

FSWEB: http://fsweb.payette.r4.fs.fed.us/units/fish.web/sediment_reports/PNF_Sediment_Monitoring_Report_1997.pdf

Nelson, R.L.; Wagoner, L.J.; Burns, D.C.; Lund, J; Faurot, M. 1996. Deposition of fine sediment in selected streams on the Payette National Forest, Idaho. Report of sediment trends and monitoring efforts, 1983-1995. Trends in cobble embeddedness, free matrix particles, surface fines in relation to mining areas in the East Fork South Fork Salmon River and Big Creek watersheds. Unpublished report. McCall, ID: U.S. Department of Agriculture, Forest Service, Payette National Forest. 145p.

FSWEB: http://fsweb.payette.r4.fs.fed.us/units/fish.web/sediment_reports/MineSed96.pdf

Nelson, R.L; McHenry, M.L.; Platts, W.S. 1991. Mining. Chapter 12 in: Meehan, W.R. 1991. Influences of forest and rangeland management on salmonid fishes and their habitat. American Fisheries Society Special Publication 19: 425-457. Bethesda, MD: American Fisheries Society.

NMFS. 2003. Endangered Species Act section 7 consultation biological opinion and Magnuson–Stevens Fishery Conservation and Management Act essential fish habitat consultation. Boise, Payette, and Sawtooth National Forest land and resource management plan revisions. Southwest Idaho Ecogroup. Ada, Adams, Boise, Camas, Custer, Elmore, Oneida, and Valley Counties, Idaho. Portland, OR: U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Northwest Region. 102p. (plus attachments).

FSWEB: http://fsweb.payette.r4.fs.fed.us/units/fish.web/biological_opinions/BO_NOAA_060903.pdf

NMFS. 1996. Making Endangered Species Act determinations of effect for individual or grouped actions at the watershed scale. Portland, OR: U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Northwest Region. 31p.

WWW: http://www.nwr.noaa.gov/1habcon/habweb/habguide/matrix_1996.pdf

NMFS. 1995. Biological opinion on implementation of interim strategies for managing anadromous fish-producing watersheds in eastern Oregon and Washington, Idaho, and portions of California (PACFISH). Silver Spring, MD: U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service. 52p.

FSWEB: http://fsweb.payette.r4.fs.fed.us/units/fish.web/documents/pacfish_bo_95.pdf

NMFS. 1993. Endangered Species Act section 7 consultation biological opinion. Reconstructing, paving, and snowplowing the South Fork Salmon River Road. Portland, OR: U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Northwest Region. 26p.

FSWEB: http://fsweb.payette.r4.fs.fed.us/units/fish.web/documents/SFSR_Road_BO.pdf

Oksanen, L. 2001. Logic of experiments in ecology: is pseudoreplication a pseudoissue? Oikos. 94:27-38.

Overton, C.K.; McIntyre, J.D.; Armstrong, R.; Whitwell, S.L.; Duncan, K.A. 1995. User's guide to fish habitat: descriptions that represent natural conditions in the Salmon River Basin, Idaho. General Technical Report INT-GTR-322. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station. 142p.

Overton, C.K.; Radko, M. A.; Nelson, R.L. 1993. Fish habitat conditions: using the Northern/Intermountain Regions' inventory procedures for detecting differences on two differently managed watersheds. General Technical Report INT-300. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station. 14p.

Overton, C.K.; Wollrab, S.P.; Roberts, B.C.; Radko, M.A. 1997. R1/R4 (Northern/Intermountain Regions) fish and fish habitat standard inventory procedures handbook. General Technical Report INT-GTR-346. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station. 73p

Platts, W.S. 1991. Livestock Grazing. Chapter 11 in: Meehan, W.R. 1991. Influences of forest and rangeland management on salmonid fishes and their habitat. American Fisheries Society Special Publication 19: 389-423. Bethesda, MD: American Fisheries Society.

Platts, W.S.; Torquemada, R.J.; McHenry, M.L.; Graham, C.K. 1989. Changes in salmon spawning and rearing habitat from increased delivery of fine sediment to the South Fork Salmon River, Idaho. Transactions of the American Fisheries Society. 118:274-283.

Platts, W.S.; Megahan, W.F.; Minshall, G.W. 1983. Methods for evaluating stream, riparian, and biotic conditions. General Technical Report INT-138. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station. 70p.

Platts, W.S.; Megahan, W.F. 1975. Time trends in riverbed sediment composition in salmon and steelhead spawning areas: South Fork Salmon River, Idaho. Transactions of the 40th North American Wildlife and Natural Resources Conference. p229-239.

Potyondy, J.P. 1988. Boise National Forest cobble embeddedness baseline inventory: results and relationship to management activities. Unpublished report. Boise, ID: U.S. Department of Agriculture, Forest Service, Boise National Forest. 37p.

Rieman, B.; Lee, D.; Burns, D.; Gresswell, R.; Young, M.; Stowell, R.; Rinne, J.; Howell, P. 2002. Status of native fishes in the western United States and Issues for Fire and Fuels Management. Forest Ecology and Management. 178:197-211.

Rieman, B.; Lee, D.; Chandler, G.; Myers, D. 1997. Does wildfire threaten extinction for salmonids? Responses of redband trout and bull trout following recent large fires on the Boise National Forest. Proceedings: first conference on fire effects on rare and endangered species and habitats, Coeur d'Alene, Idaho, November 1995. Fairfield, WA: International Association of Wildland Fire, c1997. p47-57.

Ries, R.D.; Burns, D.C.; Lund, J. 1991. Fine sediment deposition in selected tributaries to the Salmon River in the Payette National Forest. Report of monitoring results. Unpublished report. McCall, ID: U.S. Department of Agriculture, Forest Service, Payette National Forest. 33p.

FSWEB: http://fsweb.payette.r4.fs.fed.us/units/fish.web/sediment_reports/PNF_Sediment_Monitoring_Report_1991.pdf

Ries, R.D.; Burns, D.C. 1989. Embeddedness of salmonid habitat of selected streams on the Payette National Forest 1987-1988. Unpublished report. McCall, ID: U.S. Department of Agriculture, Forest Service, Payette National Forest. 36p.

FSWEB: http://fsweb.payette.r4.fs.fed.us/units/fish.web/sediment_reports/PNF_Interstitial_Sampling_Report_1989.pdf

Roper, B.B.; Kershner, J.L.; Archer, E.; Henderson, R.; Bouwes, N. 2002. An evaluation of stream habitat attributes used to monitor streams. Journal of the American Water Resources Association. 38(6):1637-1646.

Royer, T.; Minshall, G.W; Bowman, K. 1997. Assessment of habitat and biota in tributaries of Big Creek and the South Fork Salmon River, PNF. Pocatello, ID: Idaho State University, Department of Biological Sciences, Stream Ecology Center. 46p.

Royer, T.; Minshall, G.W. 1996. Habitat and biotic conditions during 1995 in streams influenced by wildfire. Pocatello, ID: Idaho State University, Department of Biological Sciences, Stream Ecology Center. 46p.

Royer, T.; Robinson, C.; Minshall, G.W. 1995. Influence of wildfire on selected streams in the PNF. Pocatello, ID: Idaho State University, Department of Biological Sciences, Stream Ecology Center. 35p.

SAS® Institute. 1990a. SAS/STAT user's guide. Version 6. Fourth edition. Cary, NC: SAS Institute Inc. 1674p. (plus appendices).

SAS® Institute. 1990b. SAS procedures guide. Version 6. Fourth edition. Cary, NC: SAS Institute Inc. 705p.

Shepard, B.B.; Leathe, S.A.; Weaver, T.M.; Enk, M.D. 1984. Monitoring levels of fine sediment within tributaries to Flathead Lake and impacts of fine sediment on bull trout recruitment. Paper presented at Wild Trout III Symposium, Yellowstone National Park, WY, September 24-25, 1984.

Sparkman, M.D. 2003. Negative influences of predacious egg-eating worms, *Haplotaxis ichthyophagous*, and fine sediments on coho salmon, *Oncorhynchus kisutch*, in natural and artificial redds. MS thesis. Arcata, CA: Humboldt State University. 55p.

Stowell, R.; Espinosa, A.; Bjornn, T.C.; Platts, W.S.; Burns, D.C.; Irving, J.S. 1983. Guide for predicting salmonid response to sediment yields in Idaho Batholith watersheds. Unpublished report. Missoula, MT, and Ogden, UT: U.S. Department of Agriculture, Forest Service, Northern and Intermountain Regions. 95p.

Suttle, K.B.; Power, M.E.; Levine, J.M.; McNeely, C. 2004. How fine sediment in riverbeds impairs growth and survival of juvenile salmonids. Ecological Applications. 14(4):969-974.

Swanston, D.N. 1991. Natural processes. Chapter 5 in: Meehan, W.R. 1991. Influences of forest and rangeland management on salmonid fishes and their habitat. American Fisheries Society Special Publication 19: 139-179. Bethesda, MD: American Fisheries Society.

Sylte, T.L. 2002. A comparison and evaluation of stream substrate embeddedness techniques. MS Thesis. Fort Collins, CO: Colorado State University, Department of Earth Resources. 122p.

Sylte, T.; Fischenich, C. 2003. An evaluation of techniques for measuring substrate embeddedness. Stream Notes. October 2003: 1-5. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Stream Systems Technology Center.

Sylte, T.; Fischenich, C. 2002. Techniques for measuring substrate embeddedness. EMRP Technical Notes Collection, ERDC TN-EMRRP-SR-36. Vicksburg, MS: U.S. Department of Defense, Army Engineer Research and Development Center. 25p.

Tagart, J.V. 1984. Coho salmon survival from egg deposition to fry emergence. In: Walton, J.M.; Houston, D.B., eds. Proceedings of the Olympic Wild Fish Conference. Port Angeles, WA: 173-181.

Tappel and Bjornn. 1983. A new method of relating size of spawning gravel to salmonid embryo survival. North American Journal of Fisheries Management. 3:123-135.

USFWS. 2003. Biological opinion for revision of land and resource management plans for the Boise, Payette, and Sawtooth National Forests. Boise, ID: U.S. Department of the Interior, Fish and Wildlife Service, Snake River Fish and Wildlife Office. 155p. (plus appendices).

FSWEB: http://fsweb.payette.r4.fs.fed.us/units/fish.web/biological_opinions/BO_USFWS_053003.pdf

USFWS. 1998. Biological opinion for the effects to bull trout from continued implementation of land and resource management plans and resource management plans as amended by the interim strategy for managing fish-producing watersheds in eastern Oregon and Washington, Idaho, western Montana, and portions of Nevada (INFISH) and the interim strategy for managing anadromous fish-producing watersheds in eastern Oregon and Washington, Idaho, and portions of California. Portland, OR and Denver, CO: U.S. Department of the Interior, Fish and Wildlife Service, Pacific and Mountain-Prairie Regions. 121p. (plus appendices).

Varner, I.M. 1948. No title. Excerpt from another document discussing an inspection by a Range Conservationist on the Forest. McCall, ID: U.S. Department of Agriculture, Forest Service, Payette National Forest. 1p.

Waters, T.F. 1995. Sediment in streams. Sources, biological effects and control. Monograph 7. Bethesda, MD: American Fisheries Society. 251p.

Wolman, M.G. 1954. A method of sampling coarse river-bed material. Transactions of the American Geophysical Union. 35(6):951-956.

Young, M.K.; Hubert, W.A.; Wesche, T.A. 1990. Effects of substrate composition and stock origin on the survival to emergence of brown trout: a laboratory study. Northwest Science. 64(4):224-231.

Zuniga, R.; Hanson, A.; Jorgensen, R.; Lesch, G.; Nelson, R. 1994. Corral burned area rehabilitation interim report. Unpublished report. McCall, ID: U.S. Department of Agriculture, Forest Service, Payette National Forest. Unnumbered pages.

Zurstadt, C. 2004. 2003 Payette National Forest fisheries–range monitoring report. McCall, ID: U.S. Department of Agriculture, Forest Service, Payette National Forest. 115p.

FSWEB: http://fsweb.payette.r4.fs.fed.us/units/fish.web/range_reports/Range_Report_2003.pdf

ACKNOWLEDGMENTS

We would like to thank several people for their assistance with this document. The following people generously donated their time to provide critical reviews of the draft report: Debbie Artimez, NOAA Fisheries Service, Boise, Idaho; Dan Duffield, US Forest Service, Ogden, Utah; Rick Henderson, US Forest Service, Logan, Utah; Jack King, US Forest Service (retired), Boise, Idaho; Bill Lind, NOAA Fisheries Service, Boise, Idaho; Don Newberry, US Forest Service, Cascade, Idaho; Bob Ries, NOAA Fisheries Service, Moscow, Idaho; Brett Roper, US Forest Service, Logan, Utah; Ally Turner, US Fish and Wildlife Service, Boise, Idaho. In addition to these, and certainly of no lesser importance, are the dozens of field crew members who labored diligently over the past three decades to collect the data used here.

Appendix 1. Glossary

Symbols

§	Section.
§§	Sections.
cfs	Cubic feet per second (ft^3/sec).
df	Degrees of freedom.
d_g	Geometric mean diameter
P	Statistical probability.

Abbreviations

BA	Biological Assessment.
BNF	Boise National Forest.
BO	Biological opinion.
CE	Cobble embeddedness.
CHAMB	Chamberlain Creek watershed (Chamberlain Basin).
EFSFSR	East Fork South Fork Salmon River watershed.
ESA	Acronym for the Endangered species Act of 1973, as amended (16 USC §§ 1531–1544)
FA	Acronym for "Functioning Appropriately" as used in the revised Forest Plan.
LOG	Base 10 logarithmic (log_{10}) transformation.
FR	Acronym for "Functioning at Risk" as used in the revised Forest Plan.
FUR	Acronym for "Functioning at Unacceptable Risk" as used in the revised LRMP.
LSFSR	Lower South Fork Salmon River watershed (*i.e.*, downstream of the mouth of the Secesh River).
LRMP	Land and Resource Management Plan.
MFSR	Middle Fork Salmon River watershed.
N	Sample size.
NFMA	National Forest Management Act.

NMFS National Marine Fisheries Service (also known as NOAA Fisheries Service).

NOAA National Oceanic and Atmospheric Administration.

PNF Payette National Forest.

RPA Forest and Rangeland Renewable Resources Planning Act (16 USC §§ 1600–1614).

SECESH Secesh River watershed.

SFSR South Fork Salmon River.

SQRT Square root transformation.

STE Survival to emergence.

U.S. United States.

USC United States Code.

USDA U.S. Department of Agriculture.

USFSR Upper South Fork Salmon River watershed (*i.e.*, upstream of the mouth of the Secesh River).

USFWS U.S. Fish and Wildlife Service.

WCI Acronym for "Watershed Condition Indicator" as used in the revised Forest Plan.

DEFINITIONS

Anadromous Characterized by a life history in which spawning and early rearing occurs in freshwater and maturation occurs in the ocean.

Mean A measure of the center of a normal distribution.

Median A measure of the center of a distribution that has the property that half of the observations are less than the specified value and half are greater; the 50^{th} percentile.

Fines Streambed sediments smaller than 6.3mm (0.25in) particle diameter; large fines.

Geometric Mean Particle Diameter The mean of a sediment particle distribution that is calculated as the square root of the product of 16^{th} and 84^{th} percentiles.

Indicator A quantifiable measure of a habitat component.

Large Fines	Streambed sediments smaller than 6.3mm (0.25in) particle diameter; fines.
Lotic	Characterized by moving waters.
Percentile	Statistical parameter indicating the percentage of a set of observations are smaller than the specified value.
Quartile	A value in a distribution that divides the distribution into four groups; first quartile = 25^{th} percentile, etc.
Salmonids	Fishes of the family *Salmonidae* (salmon, trout, and char).
Silt	Streambed sediments smaller than 0.85mm (0.033in) particle diameter; small fines.
Small Fines	Streambed sediments smaller than 0.85mm (0.033in) particle diameter; silt.

APPENDIX 2. SUPPLEMENTARY DOCUMENTS

These follow on subsequent pages. They have been slightly reduced in size to allow embedding in this report.

NELSON AND BURNS 2002

Comments on Proposed LRMP Matrix Criteria

Sediment — Chinook and Steelhead

Setting the fines (<0.85mm) standards at <12% for "functioning appropriately," 12%-20% for "functioning at risk," and >20% for "functioning at unacceptable risk" for chinook and steelhead seems too high. It is also confusing to blur the distinction between surface and depth fines in the "unacceptable risk" category. Depth fines can be adequately measured, but I'm not sure that we really measure surface fines very well; in addition, I think that they're much more variable over time and not necessarily quantitatively related to depth fines in a systematic fashion. Below is some data-based rationale for the inadequacy of the small fines criteria:

(1) Upper South Fork Salmon River Core Sampling

- The average amount of fines <0.85mm in core samples collected in 5 spawning areas from 1977 to 2000 is just 3.8%.
- The highest recorded amount of fines <0.85mm during this same period was 13.2% at Poverty Flat in 1977; however, this is the only sample that exceeded 12% in the entire period.
- The 1970s average (i.e., 1977-1979) for all 5 spawning areas was 8.4%.
- The 1980s Average (i.e., 1980-1989) for all 5 spawning areas was 5.2%.

The higher levels in the 1970s indicate that the amount of fines in the spawning areas was still decreasing following the inundation that occurred in 1965. Our monitoring indicate that this trend is continuing.

(2) Lake Creek/Secesh River Core Sampling

- The average amount of fines <08.5mm in core samples from 5 spawning areas from 1981-2000 is just 5.6%.
- One site (near Threemile Creek) has averaged 10.0% and exceeded 12% twice in the mid-1990s and is the only site that has; we have regularly identified this site as having a problem related to historic dredge mining and tailings that continue to contribute fine material.
- The 1980s average amount of fines <0.85mm (i.e., 1981-1989, with some missing years) was 5.7%.
- The 1990s average amount of fines <0.85mm (i.e., 1990-1999, with some missing years) was 5.6%.
- Excluding the Threemile Creek site, the long-term average amount of fines <0.85% in these sampling areas is just 4.5% overall and 4.3% for the 1990s.

(3) Chamberlain Basin Core sampling (Wilderness Control)

- The average amount of fines <08.5mm in core samples from 2 spawning areas from 1989-2000 (plus one sample on Chamberlain Creek from 1981) is 5.4%.
- The average for long-term average for Chamberlain Creek alone is 4.0%.

These suggest that an average amount of fines <0.85% in the 4-6% range is probably more-or-less natural.

(4) Egg-basket Embryo Survival Studies

- Studies of survival of eyed chinook eggs at Poverty Flat in 1987-88 suggest that survival approaches 0% at about 7% fines <0.85mm (see graph).

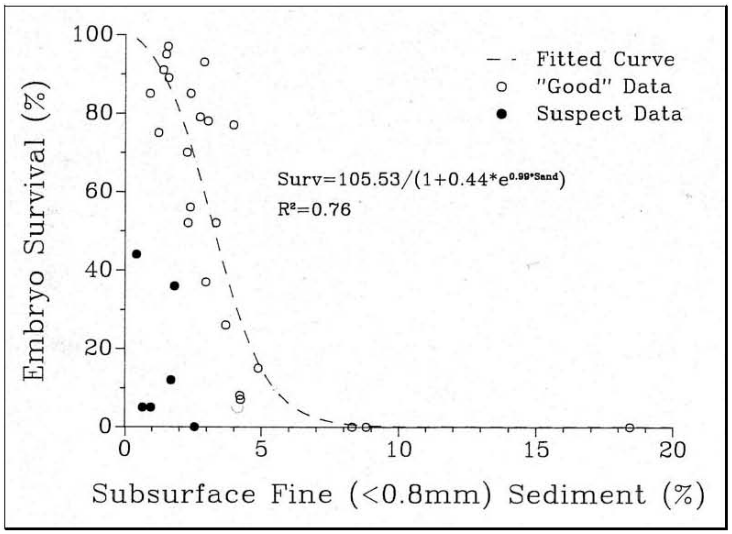

Figure 1.—Observed survival of eyed chinook eggs in egg-baskets at Poverty Flat.

- The "suspect data" identifier on this graph indicates that we felt that survival was affected by an external agent (e.g., damaged egg basket, potential predator seen in basket, etc.).
- The fact that these egg-baskets represent egg pockets but are undoubtedly different than natural egg pockets (e.g., sediment mixing is probably different, eggs have been handled, etc.) probably makes this an underestimate of the amount of fines that can be survived.
- The survival graph also suggests that there can be substantial variability in survival among redds, and that survival may be affected by relatively small increases in fines <0.85mm.

The above leads me to suggest that better breaks for defining streambed function for chinook and steelhead on the basis of instantaneous measures of fines <0.85mm would be:

- Functioning Appropriately: < 8% fines smaller than 0.85mm.
- Functioning at Risk: 8 – 11% fines smaller than 0.85mm.
- Functioning at Unacceptable Risk: >11% fines smaller than 0.85mm

Average data from several years of monitoring, which would account for natural variability and sampling error, could probably support even finer divisions. For the streams we routinely monitor, the following is probably reasonable with averaged data:

- Functioning Appropriately: < 6% fines smaller than 0.85mm.
- Functioning at Risk: 6 – 9% fines smaller than 0.85mm.
- Functioning at Unacceptable Risk: >9% fines smaller than 0.85mm

It is my opinion that surface fines should probably not be used as a criterion without better understanding of natural variability and stronger evidence that the ways in which it is measured (e.g., pebble counts and intersection-grids) are accurate (this evidence may exist and I have simply not seen it, but I am convinced that pebble counts are inaccurate).

Sediment – Bull Trout

I question whether it's reasonable to apply essentially the same criteria as chinook and steelhead to bull trout, particularly if the life history phases are not distinguished. It seems reasonable to think that smaller resident bull trout may not require substrate as large as the migratory types do, but I don't know whether there is any data to support criteria.

Temperature – Bull Trout

I think that the criteria don't reflect natural conditions very well:

- Several of our least developed streams would probably need to be classified as being at some level of risk with these criteria:
  - Monumental Creek (FCRONRW[1]) had 16 days in the second half of August, 2001, with maximum temperatures exceeding 15°C.
  - Tamarack Creek (FCRONRW) had a 7-day average maximum temperature from July to September, 1999, of 14.1°C.
  - Rapid River had a 7-day average maximum temperature from June to September, 2001, of 14.0°C.
  - Schissler Creek had a 7-day average maximum temperature from July to September, 2001, of 15.8°C.

  These are bull trout streams and probably support all life history types and life stages. It appears, also, that 15°C is too low a temperature to consider a thermal barrier to migration.

I also think that there should be some consideration of life history type (e.g., fluvial or resident) as opposed to simply life stage.

Substrate

I think there are problems with this:

[1] Frank Church River Of No Return Wilderness.

- We have at least one report suggesting revised embeddedness criteria, and it suggests much more liberal values than this matrix. We evaluated embeddedness criteria promulgated in the old Forest Plan and concluded that they were unnecessarily strict; these seem more rigid. Through 1996, the average cobble embeddedness for sites in undeveloped watersheds was 34%, which, using the proposed criteria, would identify them as predominantly "functioning at unacceptable risk."
- We also suggested that cobble embeddedness, at least as measured, was problematic because of the inability of field workers to consistently make the measurements correctly; consequently, we proposed the free matrix technique and either free matrix measurements or cobble embeddedness predicted from the relationship between free matrix and embeddedness.

Nelson and Platts (No Date)

TESTING AND REFINING THE FISHERIES-SEDIMENT RESPONSE MODEL

Summary of Preliminary Findings

Rodger L. Nelson

William S. Platts

Effective forest planning requires the ability to forecast the anticipated effects of resource development activities on all resources potentially affected by the activity, including fish populations in affected rivers and streams. Although the forces that control the abundance of animals have been the focus of ecological research for decades, the immense complexity of ecological systems has often hindered the search for environmental factors that directly limit the size of animal populations. In a resource management framework, it often becomes necessary to distill the wide array of potentially limiting factors down to a few that have some chance of being useful predictors of population changes resulting from alternative resource management scenarios. Anadromous salmonids use aquatic systems in the Idaho Batholith principally for spawning and during early growth periods (rearing). The abundance of fine sediments, both directly and as manifested by increased armoring of the streambottom (embeddedness), has been widely suggested as a potentially limiting factor for salmonid populations, and may have predictive value. In addition, fine sediment abundance and streambottom embeddedness can be quantified, and a considerable amount is known about erosion and the routing of sediments in forested watersheds.

Resource development in national forests (e.g., timber harvest and attendant road construction and vehicular traffic) often dramatically reduces the vegetal cover over relatively large tracts of land, thereby increasing potential watershed erosivity. In addition, Increased erosion may produce large amounts of sediment that will be deposited in local streams, before being transported downstream where they will influence remote aquatic ecosystems and possibly affect other economically or aesthetically valuable resources. Although disturbed areas may reproduce much of their vegetal cover, the rehabilitation process may be prolonged, and residual effects may be felt for some time. On the Boise National Forest, for example, increases in sediment yield due directly to timber harvest are thought to be insignificant 6 years after logging stops, while sedimentation caused by the roads continues (Potyondy 1988). Consequently, ecological models have been developed to predict the consequences of various resource development alternatives so that negative environmental effects can be minimized.

In 1981, a team of soil scientists and hydrologists developed a watershed sediment yield model (the R1-R4 model) (Cline et. al. 1981) to predict the potential increase in sediment yields to downstream areas that can result from proposed resource development activities. The R1-R4 was developed on the basis of the hydrologic conditions extant in the Idaho Batholith portion of the Northern Rocky Mountain physiographic province (Fenneman 1935). Even within the target area, local conditions vary considerably; consequently, the various National Forests have developed their own variants of the R1-R4 model, tailored specifically to the hydrologic conditions that prevail in their area of interest. Thus, the Nez Perce National Forest in northern Idaho relies upon a local variant named "NEZSED", whereas the Boise National Forest uses the "BOISED" model.

Because many of the streams draining the Idaho Batholith are or have been important spawning and rearing streams for anadromous fish, particularly chinook salmon (*Oncorhynchus tshawytscha*) and steelhead trout (*O. mykiss*)[1], as well as a variety of resident salmonid fishes, potential effects of forest development on salmonid populations and their habitat is a principal area of concern. Consequently, a group of fisheries biologists developed the Fisheries-Sediment Response Model (hereafter referred to as FISHSED1) to translate expected increases in sediment delivery (as predicted by the R1-R4 model) into potential loss of salmonid fish habitat and consequent reductions in population densities and spawning success by, principally, anadromous chinook salmon and steelhead trout (Stowell et al., 1983). Stowell et al. (1983) provide guidelines for the application of FISHSED1, and warn users that the relationships used are general and possibly too simplistic for use in some areas. As with the R1-R4 model, specific functional relationships used in FISHSED1 may be modified to suit local conditions. The original model is presently available as part of the U.S. Forest Service's General Aquatic Wildlife System (GAWS).

Although FISHSED1 was developed primarily for aquatic systems containing chinook salmon or steelhead trout, it has been applied to other areas as well in conjunction with the development of forest plans. In addition, the original formulation was largely an intuitive model that represented the experiences and opinions of its various authors, and has been inadequately tested under real-world conditions. This study takes the first long, hard look at the ability of FISHSED1 to predict salmonid habitat and population responses by using actual sediment delivery predictions submitted by National Forest System soil scientists and hydrologists, proposes modified relationships and alternative ways of interpreting the effects of anticipated changes in sediment yields on aquatic habitat, and identifies areas where additional research is needed.

CRITICAL ASSUMPTIONS OF FISHSED1

Like all mathematical attempts to model the "real world," the adequacy of FISHSED1 is largely dependent upon the validity of its assumptions. In addition, because FISHSED1 uses the predictive output of the R1-R4 model as its basic input, it also depends upon the assumptions used in the formulation of the R1-R4 model and its geographic variants. Since both of these models deal with very complex and poorly

[1]*Oncorhynchus mykiss* has just recently been approved as the correct binomial name for steelhead and rainbow trout by the American Fisheries Society. These commercially and culturally valuable fish were previously designated (*Salmo gairdneri*), but recent studies have determined that they belong in the same genus as Pacific salmon and not in that of Atlantic salmon and brown trout (a European species), and that they and the Kamchatkan trout of eastern Asia are conspecific. Since the former trout was described first, its binomial name has been applied to all populations of these geographically widespread fish.

understood processes, they are largely conceptual and reflect not only the knowledge of the expert committees that developed them, but, inevitably, their baises as well. Consequently, the entire process of attempting to predict impacts to fisheries from proposed forest development activities is essentially one of expert guesswork based upon more expert guesswork. Despite this limitation, however, they represent the best tools currently available for assisting with forest planning so that environmental risks can be minimized. Even inadequate tools are better than no tools.

Sediment Disposition within the Critical Reach

Accumulation

The R1-R4 model provides the basic input to FISHSED1, predicted sediment yield over natural rates delivered to a critical reach. If this figure is inaccurate, FISHSED1 cannot be expected to accurately predict fisheries response. It is not the purpose of this study to evaluate the accuracy of the R1-R4 model, we must assume it is correct. However, the value itself, sediment yield over natural, may not be sufficient for modelling fisheries response; some indication of what happens to sediment once it is delivered to the critical reach should be included. In fact, Cline et al. (1981) suggest that the question of sediment disposition is the weakest link in predictions of fishery impacts using R1-R4 Model predictions.

Recovery

Under natural conditions, most fines delivered to the critical reach will be flushed from the system. Above some threshold input level, however, they may begin to accumulate in the system, particularly in low velocity microhabitats. Similarly, as the watershed recovers following a resource development activity, sediment delivery to the critical reach will be reduced, and accumulated sediments may begin to be cleaned from the system. This process is clear in concept (Figure 1), but the value of the hypothesized threshold sediment yield is obscure.

Since FISHSED1 contains no provision for dealing with sediment disposition once it is routed to the critical reach, either as accumulation or as recovery, application of FISHSED1 to sediment yield predictions is, at this time, likely to be based on single year or successive single year (increasing annual yields) predictions of sediment yield, a procedure that would be consistent with no storage of excess sediments between entries. Should sediment yields be greater than the threshold level, however, sediments may be retained in the critical reach and FISHSED1 predictions may, if the rest of the modelled relationship is accurate, underestimate actual sediment levels and overestimate fishery potential.

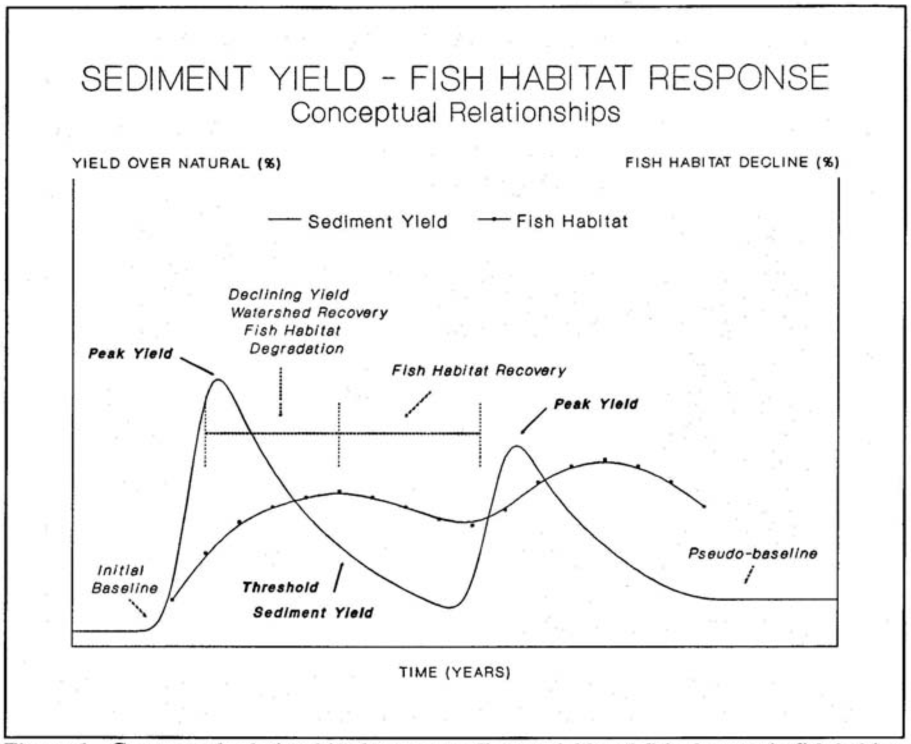

Figure 1.--Conceptual relationships between sediment yield and fish changes in fish habitat quality.

FINE SEDIMENT AS A LIMITING FACTOR

FISHSED1 was designed to deal with the effects of increased fine sediment on fish populations subject to the provision that such sediment was the most likely factor limiting salmonid populations in the critical reach. Stowell et al. (1983) suggest that the manager faced with the possible need to use the model "should evaluate all factors and determine if changes in sedimentation will affect the abundance of fish or capacity of the habitat to support fish" (p.10). The present concern with incresed sediment production resulting from forest management activities, the reltive ease with which sedimentation can be measured, and the large body of evidence that implicates it as an important habitat factor

5

for salmonid populations leave the door open for disregarding other potentially important influences on salmonid populations.

Embryo Survival and Emergence

Information about the intra-gravel requirements of developing salmonids is currently far from complete. It is widely accepted among fisheries biologists, and well supported in the literature, that excessive amounts of fine sediments in the egg pocket can kill developing embryos by restricting the intra-gravel flow that is needed to deliver dissolved oxygen and remove waste products. Several studies (e.g., McCuddin 1977; Tappel and Bjornn 1983; Irving and Bjornn 1984) have been reported that support the existence of a negative relationship between fine sediments in spawning gravels and embryo survival and fry emergence. However, these represent laboratory studies that use a simulated redd with "simplified, unnatural gravel mixtures" (Everest et al. 1987) that may bear little resemblance to natural redds. Even more to the point, Chapman and McLeod (1987) were referring to the laboratory studies used in FISHSED1 when they stated that "[i]t is always inappropriate to extrapolate such information to field conditions."

Particle Sizes

In addition to the question of extrapolating laboratory studies to field situations there is the lack of agreement among researchers as to what particle sizes constitute fine sediments; which size categories of fines are critical and what developmental phases they impact most are obscure. Clearly, the smaller the material the more likely it is to restrict intra-gravel flow, and the smaller fines may therefore be the most destructive to developing embryos. Fines that are too large to significantly retard intra-gravel flow, however, may still be too small to permit larvae to move upward in the redd and emerge from the gravel. Studies have commonly used sediments of 9.5 mm and smaller, 6.3mm and smaller, 4.8mm and smaller, and 0.85mm and smaller. FISHSED1 is based on studies that used sediments less than 6.3mm in diameter; however, sediments of 4.8 to 6.3mm have been counted as gravel rather than fine sediments elsewhere (e.g. Platts et al. 1983). If the fine sediments limiting embryo survival were actually smaller than 4.8mm (or even 0.85mm), then using the percentage of 6.3mm and smaller sediments as a criterion should introduce a large amount of variability into the relationship between fine sediment and survival of embryos.

Winter Carrying Capacities

The relationship between aquatic habitat condition and winter carrying capacity (i.e. the density of fish that can be supported by the habitat through the winter) is presently an active research arena. FISHSED1 assumes that rubble bottoms of pool habitats are key areas for over-wintering salmonids, and laboratory-derived relationships from Bjornn et al. (1977) are used to predict changes in fish density in response to changes in the embeddedness of this habitat. Use of these relationships requires two fundamental assumptions: first, that embeddedness is either the principal factor limiting over-wintering capacity to support juvenile salmonids, or is an accurate surrogate for the actual limiting factor; and, secondly, that laboratory findings can be extrapolated directly to natural situations.

As embeddedness increases, the interstitial space in the streambottom substrate decreases, thereby reducing the extent and quality of these protective refuges in which salmonid fry can escape winter conditions. There is no obvious reason to question this relationship in a qualitative sense, but there is evidence that embeddedness may not be the only factor that determines winter carrying capacity. Hillman et al. (1987) demonstrated that placing rubble piles in potential in-channel wintering areas initially increased the number of chinook salmon fry that began the winter there; however, the effect of the rubble piles was greatest when located near such areas of cover as undercut streambanks. Similarly, Swales et al. (1986) associated successful over-wintering habitat for chinook salmon that remained in small interior rivers with deep pools containing large debris, and for coho salmon with ponds and pools that contained organic cover.

The question of extrapolating laboratory studies to field situations exists here also; similar cautions are required with predictions of winter carrying capacity based on embeddedness. Chapman and McLeod (1986) devote several pages to criticism of such extrapolation, and have even singled out the use of relationships from Bjornn et al. (1977) in FISHSED1. Their principal objection to using the relationships of Bjornn et al. (1977) in FISHSED1 is the abnormally high density of fish used in the artificial channels and the correspondingly high densities that the relationships would predict at low embeddedness. Chapman and McLeod (1987) contend that this result may be due to excessively high stocking densities in the artificial channels. Whatever the cause, fish densities predicted by FISHSED1, in response to virtually any embeddedness level, are too high for even undisturbed streams. Inspection of these relationships (Stowell et al. 1983, p. 70) suggests that densities as high as 16 fish/m^2 (age 0 cutthroat trout) are reasonable at 0% embeddedness, falling off to about 4 fish/m^2 at 100% embeddedness. A recent survey of fish densities in western streams (Platts and McHenry 1988) lists no instance of cutthroat trout densities over 2.51 fish/m^2 in either the Pacific of Columbia Forest ecoregions; most densities were much lower.

Use of the embeddedness-fish response curves of Bjornn et al. (1977) in FISHSED1 was modified to allow determination of percent of natural density as a hedge against the high laboratory densities. Whether these percentages are reliable is unknown and largely untestable. Natural carrying capacities of streams is not well known, so only actual densities can be tested in this study. Monitoring fish densities before and after a land-use activity may allow the percentage change predictions of FISHSED1 to be tested, but in the meantime we can only extrapolate from observed densities and embeddedness levels and estimate whether the predictions at natural embeddedness seem reasonable.

7

Summer Carrying Capacities

Stowell et al. (1983) suggests that rubble bottoms of glide habitats are key areas for summer rearing of juvenile salmonids, and FISHSED1 incorporates embeddedness as the key factor limiting summer rearing capacity. FISHSED1 relies on relationships between embeddedness and summer rearing capacity that were developed by laboratory studies of Bjornn et al. (1977), and requires two fundamental assumptions: first, that embeddedness is the principal factor limiting summer capacity to support juvenile salmonids, or is an accurate surrogate variable; and, second, that laboratory studies can be extrapolated directly to natural situations. The same cautions mentioned with respect to winter carrying capacity apply here.

Determining Embeddedness

Stowell et al. (1983) do not specifically state the criteria or techniques that should be used with respect to embeddness measurements when applying FISHSED1, and either simple ocular estimation (Platts et al. 1983) or the hoop embeddeness measurement techniques developed by Kelley and Dettman (1980) with the criteria of Burns (1984) and Burns and Edwards (1985) are frequently used. The latter methodology is very costly in terms of human effort, tends to be restricted to pre-selected instream habitat types rather than being reflective of overall reach conditions, and may pose statistical problems because of the small numbers of independent samples that are frequently taken. While hoop embeddedness offers the apparent security of being a "measured" value rather than one based on observer estimation, the results of Torquemada and Platts (1988) provide little incentive for using it rather than several less costly techniques. Testing, to be discussed later, used measures of embeddedness based on both the hoop and ocular (Platts et al. 1983) techniques, but bases conclusions principally upon the latter.

Channel Classification

The FISHSED1 model relies on channel classifications following Collotzi (1974), Espinosa et al. (1981), and Rosgen and Silvey (1980). These classifications stratify stream channels into 3 broad categories based on width, depth, velocity, discharge, gradient, bed roughness, bed material particle size, and sinuosity. For FISHSED1, three principal channel types are recognized: A-channels, with high gradients, narrow, well incised "v"-shaped canyons; B-channels, with moderate gradients, broad, moderately incised "v"-shaped canyons; and C-channels, with low gradients, broad, poorly incised channels. (See Appendix C of Stowell et al. [1983] or the cited references for more thorough descriptions of channel types.)

The significance of channel classification to FISHSED1 lies in the differences in sediment transport and storage in the different channel types. Sediment transport competence decreases with channel type, with A-channels being the most competent and

C-channels the least. Consequently, sediment yields over natural from surrounding watersheds will probably be felt most strongly in C channels, which also happen to be the most likely channels to support anadromous fish spawning areas. The FISHSED1 model uses calculated relationships between embeddedness and sediment delivery over natural and between depth fines and sediment delivery over natural to predict in-channel fish habitat responses to increases in sediment yield. Although the functional relationships used reflect observational data, it is not clear whether they incorporate cumulative effects of sustained above-natural sediment yield or whether they are merely point estimates. This consideration is analogous to that discussed for sediment disposition and is a potential source of error. The accuracy with which FISHSED1 can predict changes in embeddedness and depth fines is obviously dependent on the validity of these functional relationships. These relationships can be directly tested by comparing model predictions with actual habitat conditions.

METHODS

Testing Areas

FISHSED1 was tested in two disjunct drainages in the Northern Rocky Mountain Physiographic Province (Fenneman 1931) in the state of Idaho. In the Clearwater River drainage, study areas were established on the North Fork of Red River (NFRR), the South Fork of Red River (SFRR), on Red River (RR) below the confluence of NFRR and SFRR, and on Trapper Creek (TC), a tributary of SFRR. In the Salmon River drainage, study areas were established at 4 locations on the South Fork Salmon River (SFSR) at Upper Stolle Meadows (USM), Lower Stolle Meadows (LSM), and Poverty Flat (PF); an additional study area was located in Tyndall Meadows (TM) on Johnson Creek, a major SFSR tributary.

Each study area contained a pair of study sites to allow replication. The replicates were designated upper or lower in relation to streamflow; thus, SFRR contained two sites, Upper SFRR (USFRR) and Lower SFRR (LSFRR). Throughout this document, sites will be referred to by their acronyms. In the Clearwater study areas, each site spanned 100 m of stream and contained 33 transects, placed at 10-m intervals and perpendicular to the apparent average streamflow. The Salmon River study areas were older and part of concurrent studies, and were therefore of more variable design. In most cases, each study site spanned 183 m of stream and contained 60 or 61 transects; PF was an exception with each site spanning 49 m and containing 16 transects.

Egg to Alevin Survival

In mid-October of 1986 and 1987, eyed chinook salmon eggs at a developmental stage of approximately 150 temperature units (TUs) were obtained from the McCall, Idaho, salmon hatchery, for planting in the PF study area. In 1986, an additional lot of eggs was obtained from the Rapid River hatchery near Riggins, Idaho, for planting in the NFRR study area. This procedure insured that eggs planted in the two study areas were from salmon stocks indigenous to that river system.

Egg baskets for use in 1986 were constructed of plastic netting with an 0.6-cm mesh size, and were used without tops. In 1987, reusable baskets were covered with plastic netting of 0.4-cm mesh and fitted with tops; new baskets were constructed exclusively of 0.4-cm mesh. On site, streambottom substrate materials were excavated from apparent redds, placed in the egg baskets in about their natural proportions, and eggs were deposited into the centrum of this artificial egg pocket by way of a small PVC tube. Individual baskets were then placed in inside reinforcement bar frames, replaced in the hole from which the substrate material had been removed, and covered with loose material. In 1986, 20 baskets were placed in each study area (PF and NFRR). Although most baskets contained 150 eggs, the actual number of eggs per basket ranged from 89 to 150. In 1987, 40 baskets were placed in the PF study area and all contained 150 eggs.

Egg baskets were recovered in April of each year. Basket contents were placed in trays and viable alevins were counted. The sediment material was then brought back into the laboratory, oven dried, and shaken through USA Standard Testing Sieves to determine the dry weight proportions of each sediment class per basket.

Sediment Analysis

Depth Fines

Depth (subsurface) fines were sampled by obtaining core samples. Sediments within each core were excavated to a depth of approximately 30 cm (less if the substrate could not be penetrated) using a McNeil-type coring tube. A total of 6 core samples from each study site was intended, but fewer were actually obtained in some cases. In the laboratory, the core contents were oven dried, and shaken through USA Standard Testing Sieves to determine the dry weight proportions of each sediment class per basket. Depth fines amounts for the Salmon River drainage sites, where available, were obtained from the Boise National Forest.

Embeddedness

Embeddedness was evaluated in two ways. In 1984 and 1985 it was measured using the hoop technique of Kelley and Dettman (1980); in 1984, however, only microhabitats meeting the criteria of Burns (1984) and Burns and Edwards (1985) were sampled, whereas samples were taken from each of the three major microhabitat types (pool, run or glide, and riffle) in 1985[2]. Embeddedness was also measured along each transect in each of the study sites using the ocular technique of Platts et al. (1983). Using both techniques allows comparison between methods and evaluation of which is most satisfactory for modelling purposes.

Stream Size

Only two stream size statistics were of interest, stream width and length. Length was taken as the length of the study site, whereas width was taken as mean stream width in the study site. Stream width was measured at each transect according to the methodology of Platts et al. (1983), summed, and divided by the number of transects to obtain average width.

Fish Population Analysis

Fish populations were sampled by electrofishing, using the four-pass removal-depletion method of Platts et al. (1983). Salmonids were all identified to species, counted, measured to the nearest 1 mm, weighed to the nearest 0.1 gm, and returned live to the stream well downstream of the point of capture. Populations were sorted by age class (determined by length-frequency plots), and sizes of the young-of-the-year class were estimated using the maximum-likelihood model of Platts et al. (1983). Estimated density ($\hat{D}$) was calculated as:

$$\hat{D} = \hat{N} * \bar{w} * \bar{L} \qquad (1)$$

Where: $\hat{N}$ is the estimated population size, $\bar{w}$ is the average stream width, and $\bar{L}$ is the study site length. Mean fish weight was calculated as the sum of the individual weights divided by the actual number of fish collected.

Non-game species were identified at least to genus, counted, weighed in mass, and returned live to the stream well below the area where sampling was being performed.

[2]Because of limitations of the sampling technique, sampling in all microhabitats led to some difficulty. Principal problems were that cobble embeddedness cannot be readily measured with this technique on a substrate of all fine sediments (see Torquemada and Platts 1988) and pools over arm's-length in depth prevented sampling some of these microhabitats.

Model Application

Two approaches were used to test FISHSED1, a non-cumulative and a cumulative approach. Estimates of sediment yield (R1-R4 model output) for the two drainages were obtained from hydrologists on the Nez Perce and Payette National Forests.

Non-cumulative Approach

The non-cumulative approach assumed that a sediment-producing event on the watershed was essentially independent of succeeding events of lesser magnitude; consequently, it ignored any consequences relating to watershed recovery. Thus, if a sediment producing event of some magnitude were followed by an event of greater magnitude, both events were considered to have contributed to off-site fish habitat degradation. If, however, the first event were followed by an event of lesser magnitude, only the first event would have been considered to have contributed to downstream habitat degradation. If streams can adequately flush sediments routed to them from the first event before excess sediments are delivered by the succeeding event, the assumption would be valid. This is the simplest application of FISHSED1, and the one most likely to be routinely employed; it is also the approach most thoroughly examined in this report.

Cumulative Approach

The cumulative approach assumed that sequential sediment yield events were not independent until a developed or pseudo-baseline yield was developed. This approach incorporated the effect of delayed watershed recovery, though it continued to ignore any effects of aquatic habitat recovery. Thus, the two scenarios described above would each have had two events that contributed to downstream degradation. If streams could not flush themselves between successive events, even if succeeding events were smaller than preceding events, then this assumption would be valid. This approach is somewhat more complicated than the non-cumulative approach, but not excessively so. This approach intuitively seems more reasonable than the non-cumulative approach, because sediment deposited by watershed events may be stored in certain low gradient reaches despite their rapid removal from higher gradient reaches.

Model Refinement

At this stage, model refinement consisted of exploring different mathematical relationships for the functional relationships used in FISHSED1. These included relationships between trout and salmon density and embeddedness, embeddedness and abundance of fines, surface and depth fines, large and small depth fines, and chinook egg to alevin survival for both large and small fines. Linear and non-linear curves were fit using PlotIT® statistical graphics software. Two new microcomputer programs are currently being developed, one in Microsoft® FORTRAN77 (for portability to the USFS

12

Data General System) and one in Microsoft® QuickBASIC™ (for user-friendly convenience).

PRELIMINARY RESULTS

Non-Cumulative Approach

Trapper Creek

Using the non-cumulative approach, all FISHSED-1 physical habitat and fish density predictions for TC except depth fines appear unrealistic (Table 1).

Table 1.--Comparison of observed cutthroat trout population and habitat conditions with FISHSED1 predictions (non-cumulative approach), Trapper Creek, Nez Perce National Forest, Idaho. All values except carrying capacities and densities are expressed as a percentage; carrying capacities and densities are expressed as fish/m^2.

	Predicted Values			Observed Values			
			Summer Rearing	Embeddedness			Summer
Year	Embedded	Fines	Capacity	Ocular	Hoop	Fines	Density
			Upper Trapper Creek				
1984	24.0	19.6	9.64	60 [1]	14.5	21	0.22
1985	24.1	19.7	9.63	55.6	34.5	NA	0.12
1986	23.9	19.5	9.64	54.8	NA	NA	0.10
Avg.	24.0	19.6	9.64	56.8	24.5	21	0.15
			Lower Trapper Creek				
1984	24.0	19.6	9.64	55 [1]	14.5	21	0.16
1985	24.1	19.7	9.63	47.0	34.5	NA	0.13
1986	23.9	19.5	9.64	50.7	NA	NA	0.11
Avg.	24.0	19.6	9.64	50.9	24.5	21	0.13

[1] Converted from older embeddedness ranking scale.

Stream ocular embeddedness was much higher than predicted embeddedness, with an average difference of 124%. Even hoop embeddedness for 1985 was 75% higher than the corresponding FISHSED-1 prediction.

13

Depth fines were only measured in 1984, and actual values were quite similar to predicted values.

Actual summer cutthroat trout fry densities were far below the steelhead fry densities predicted by FISHSED1. The model predicted steelhead fry capacities to range from 9.63 to 9.64 fish/m^2, or 96.3 to 96.4% of full capacity. Since Trapper Creek contains only cutthroat trout, which are not considered in FISHSED1 with respect to summer densities, we have to assume that steelhead and cutthroat fry are similarly responsive to embeddedness; given the similarities of the two species, this assumption seems adequate in this case. Actual densities were 0.12 and 0.11 fish/m^2 for 1985 and 1986, respectively, nearly ***two orders*** of magnitude below predicted densities. Platts and McHenry (1988) report trout densities from several Idaho streams, and averaging their reported values yields a density of 0.10 fish/m^2, approximately the densities observed on Trapper Creek.

South Fork Red River

Actual aquatic habitat conditions for the SFRR sites were generally inconsistent with FISHSED1 predictions (Table 2), but the disparities were generally less than for TC.

Table 2.--Comparison of observed fish habitat conditions with FISHSED1 model predictions (non-cumulative approach), South Fork Red River, Nez Perce National Forest, Idaho. All values except carrying capacities are expressed as a percentage; carrying capacities are expressed as fish/m^2.

	Predicted Values				Observed Values						
			Summer Rearing Cap.		Embeddedness			Summer Fish Density			
Year	Embedded	Fines	Steel.	Chin.	Ocular	Hoop	Fines	Steel.	Cutt.	Chin.	Total
					Upper South Fork Red River						
1984	24.2	19.8	9.63	1.53	48[1]	21.4	NA	0.00	0.02	0.13	0.15
1985	24.2	19.8	9.63	1.53	41.2	33.2	36.7	0.01	0.04	0.00	0.05
1986	24.2	19.8	9.63	1.53	34.7	NA	NA	0.01	0.04	0.00	0.05
Avg.	24.2	19.8	9.63	1.53	41.3	27.3	36.7	0.01	0.03	0.04	0.08
					Lower South Fork Red River						
1984	24.2	19.8	9.63	1.53	63[1]	21.4	NA	0.00	0.03	0.04	0.07
1985	24.2	19.8	9.63	1.53	42.9	33.2	36.7	0.01[2]	0.06	0.01	0.08
1986	24.2	19.8	9.63	1.53	37.2	NA	NA	0.00	0.04	0.00	0.04
Avg.	24.2	19.8	9.63	1.53	47.7	27.3	36.7		0.04	0.02	0.06

[1] Converted from older embeddedness ranking scale.
[2] One fish collected.

Actual aquatic habitat conditions for the SFRR site were generally inconsistent with FISHSED1 predictions (Table 2), but the disparities were generally less than for TC.

14

Ocular embeddedness ranged from 43% (USFRR, 1986) to about 163% (LSFRR, 1984) greater than FISHSED1 predictions. Restricting ocular embeddedness measures to glide habitat had little effect on the magnitudes of the differences between actual and predicted levels. Hoop estimates were closer at 12% lower and 37% greater than FISHSED1 predictions for 1984 and 1985, respectively. Again, however, the hoop estimate that was closer to the model prediction was based on only one hoop in an unspecified habitat type; it should therefore be viewed with caution. In addition, the 1985 hoop estimates undersampled pool habitats (only one pool sampled); including the single pool sample twice in the overall average would raise the average to 39.2%, very close to the corresponding ocular measurement. Ocular embeddedness declined in both SFRR sites from 1984 to 1986.

The single core sample estimate of depth fines (1985) was 85% greater than predicted by FISHSED1.

Summer carrying capacities for steelhead trout fry predicted by FISHSED-1 were more than two orders of magnitude greater than actual steelhead trout fry densities. In fact, steelhead and cutthroat fry together made up, on average, only 0.6% of the FISHSED1 prediction for steelhead fry; even with chinook added to the total, actual densities were far below FISHSED1 predictions. Since cutthroat should exhibit higher densities (according to response curves in Appendix E of Stowell et al. (1983)), over-prediction by the model may be even more severe than the data suggest at first glance. Average density for 2 rainbow trout streams reported by Platts and McHenry (1988) was 0.11 fish/m^2, 38 and 120% greater, for 1985 and 1986, respectively, than average densities observed on SFRR.

Chinook occurred sporadically in the SFRR sites, but when present they were generally found to be at a densities about one order of magnitude below FISHSED-1 predictions. The really interesting thing is that their average density was higher than for either steelhead or cutthroat, both of which FISHSED-1 always predicts to be more abundant.

North Fork Red River

In the NFRR study area, FISHSED1 predictions were invariably lower than actual habitat conditions (Table 3).

Ocular embeddedness estimates ranged from 85% (1986) to 166% greater than predicted values. Hoop estimates were closer to predicted values than ocular estimates, but they did not represent all habitat types (1984 types were unspecified while 1985 samples did not include any pool). Ocular embeddedness levels in glide habitats only were closer to FISHSED1 predictions (89 and 71% different for 1985 and 1986 [averaged for both study sites], respectively) and were within 10 percentage points of hoop estimates. Both embeddedness measurement techniques were lower in 1985 than in 1984, and ocular embeddedness declined further in 1986; the rate of decline in ocular embeddedness was 7.5% per year.

Table 3.--Comparison of actual fish habitat conditions and population densities with FISHSED1 model predictions (non-cumulative approach) North Fork (Upper) Red River Nez Perce National Forest, Idaho. All values except carrying capacities are expressed as percentage; carrying capacities are expressed as fish/m^2.

	Predicted Values						Actual Values						
			Rearing Capacity				Embeddedness			Fish Density			
					Chinook							Chinook	
Year	Embed.	Fines	Steel. Summer	Cutt. Summer	Winter	Summer	Ocular	Hoop	Fines	Steel. Summer	Cutt. Summer	Winter	Summer
							Upper North Fork Red River						
1984	27.4	20.2	9.52	NA	1.47	2.39	65 [1]	42.0	42.4	0.00 [2]	0.01	0.15	0.59
1985	27.4	20.2	9.52	NA	1.47	2.39	62.0	36.5	37.7	0.00	0.01	0.09	0.75
1986	27.4	20.2	9.52	NA	1.47	2.39	50.7	NA	NA	0.03	0.01	NA	0.93
Avg.	27.4	20.2	9.52	NA	1.47	2.39	59.2	39.3	40.1	0.01	0.01	0.12	0.76
							Lower North Fork Red River						
1984	27.4	20.2	9.52	NA	1.47	2.39	73 [1]	42.0	42.4	0.01	0.02	0.15	0.47
1985	27.4	20.2	9.52	NA	1.47	2.39	71.0	36.5	37.7	0.04	0.00 [3]	0.09	0.54
1986	27.4	20.2	9.52	NA	1.47	2.39	64.6	NA	NA	0.01	0.02	NA	1.08
Avg.	27.4	20.2	9.52	NA	1.47	2.39	69.5	39.3	40.1	0.02	0.01	0.12	0.70

[1] Converted from older embeddedness ranking scale.
[2] One fish collected.
[3] Two fish collected.

Measured amounts of depth fines were 110 and 87% greater than FISHSED1 predictions in 1984 and 1985, respectively; like embeddedness, fine sediments declined (by 11.1%) between 1984 and 1985. FISHSED1 predictions of chinook fry summer rearing capacity were extremely high compared to actual salmon densities. The greatest difference (LNFRR, 1984) was a prediction 409% higher than actual, but even the closest prediction (LNFRR, 1986) was 121% too high.

Chinook densities were, however, far higher than observed densities of steelhead and cutthroat trout fry in the other Red River drainage study areas, a situation that contradicts the densities expected from the response curves in Appendix E of Stowell et al. (1983). Steelhead and cutthroat trout occurred in the NFRR sites at densities about two orders of magnitude below capacities predicted by FISHSED1.

16

Red River

FISHSED1 predictions were invariably lower than observed habitat conditions (Table 4).

Ocular embeddedness ranged from 79% (URR, 1985) to 137% (LRR, 1986) higher than model predictions. Ocular measurements of embeddedness in glide habitat alone (43.5 and 55.1% for 1985 and 1986 [averaged for both sites], respectively) were closer to model predictions, though they were within 10 percentage points of overall reach estimates. The only available hoop measurement of embeddedness (1985) was higher than model predictions by 100%, but included habitats not conforming to the criteria of Burns and Edwards (1985). When only samples fitting these criteria were considered, hoop embeddedness was estimated at 32.9%, only 20% greater than the FISHSED1 value.

Actual abundances of depth fines exceeded FISHSED1 values by 52 and 77% in 1984 and 1985, respectively.

Table 4.--Comparison of actual fish habitat conditions and population densities with FISHSED-1 model predictions (non-cumulative approach), Red River (below Red River R.S.), Nez Perce National Forest, Idaho. All values except fish densities and carrying capacities are expressed as a percentage; carrying capacities are expressed as fish/m^2.

	Predicted Values		Actual Values		
			Embeddedness		
Year	Embedded	Fines	Ocular	Hoop	Fines
			Upper Red River		
1984	27.4	20.2	58 [1]	NA	30.7
1985	27.4	20.2	49.0	54.8	35.8
1986	27.4	20.2	64.1	NA	NA
Avg.	27.4	20.2	57.0	54.8	33.3
			Lower Red River		
1984	27.4	20.2	65 [1]	NA	30.7
1985	27.4	20.2	51.1	54.8	35.8
1986	27.4	20.2	60.8	NA	NA
Avg.	27.4	20.2	59.0	54.8	33.3

[1] Converted from older embeddedness ranking scale.

Upper Stolle Meadows

Based on the sediment yield input predictions of BOISED model (the Boise National Forest version of the R1-R4 model), FISHSED1 predictions of embeddedness

17

in the Upper Stolle Meadows study sites were far to low (Table 5). Actual ocular embeddedness exceeded model predictions by 122 (Upper Site, 1986) to 200% (Upper Site, 1984). In addition, embeddedness declined in the Upper Site by 26% during the study period, which could not be accounted for by either FISHSED1 or the R1-R4 model.

Table 5.--Comparison of actual bull trout population densities and habitat conditions with FISHSED1 model predictions (non-cumulative approach) for steelhead trout, Upper Stolle, Upper Site, Boise National Forest, Idaho. All values except fish densities and carrying capacities are expressed as a percentage; carrying capacities are expressed as a fish/m^2.

	Predicted Values			Actual Values			
				Embeddedness			
Year	Embedded	Fines	Summer Rearing Capacity	Ocular	Hoop	Fines	Summer Density
				Upper Site			
1984	23.3	18.6	9.8	70 [1]	NA	NA	0.12
1985	23.3	18.6	9.8	54.1	NA	NA	0.10
1986	23.3	18.6	9.8	51.7	NA	NA	NA
Avg.	23.3	18.6	9.8	58.6	NA	NA	0.11
				Lower Site			
1984	23.3	18.6	9.7	63 [1]	NA	NA	0.12
1985	23.3	18.6	9.7	61.0	NA	NA	0.17
1986	23.3	18.6	9.7	65.9	NA	NA	NA
Avg.	23.3	18.6	9.7	63.3	NA	NA	0.15

[1] Converted from older embeddedness ranking scale.

Fines data are not yet available for these sites.

FISHSED1 was not originally designed to include bull trout, the only indigenous species in the Upper Stolle Meadows area. Although this portion of the SFSR supports a healthy population of bull trout, predictions of summer fry density using FISHSED1's steelhead trout relationships were much higher than actual trout densities. The FISHSED1 prediction of 9.7 fish/m^2 was an astounding 7,362% higher than the actual combined average trout density of 0.13 fish/m^2 in the Upper Stolle Meadows sites.

Tyndall Meadows

FISHSED1 predictions of embeddedness in the Tyndall Meadows study sites were far too low (Table 6). Actual ocular embeddedness was 167 and 198% higher in the upper and lower sites, respectively, than predicted by FISHSED1.

Data on depth fines are not available for these sites yet.

Table 6.--Comparison of actual fish population densities and habitat conditions embryo survival, and rearing capacity with FISHSED-1 model predictions (non-cumulative approach), Upper Tyndall Meadows, Boise National Forest, Idaho. All values except fish densities and carrying capacities are expressed as a percentage; carrying capacities are expressed as a fish/m^2.

	Predicted Values			Actual Values			
Year	Embedded	Fines	Summer Rearing Capacity	Embeddedness: Ocular	Embeddedness: Hoop	Fines	Summer Density
			Upper Tyndall Meadows				
1984	23.7	19.1	9.6	NA [1]			NA
1985	23.7	19.1	9.6	63.2			0.51
1986	23.7	19.1	9.6	NA			NA
Avg.	23.7	19.1	9.6	63.2			0.51
			Lower Tyndall Meadows				
1984	23.7	19.1	9.6	NA [1]			NA
1985	23.7	19.1	9.6	70.7			0.32
1986	23.7	19.1	9.6	NA			NA
Avg.	23.7	19.1	9.6	70.7			0.32

[1] Converted from older embeddedness ranking scale.

FISHSED1 was not originally designed to include brook trout, the dominant species in the Tyndall Meadows area. Although this portion of the Johnson Creek supports a healthy population of brook trout, predictions of summer fry density using FISHSED1's steelhead trout relationships were much higher than actual trout densities. The FISHSED1 prediction of 9.6 fish/m^2 was a whopping 2,186% higher than the actual combined average trout density of 0.42 fish/m^2 in the Tyndall Meadows sites.

Non-Cumulative Approach

Tseting of the non-cumulative approach is incomplete at this time. It appears, however, that it may fit TC better than the non-cumulative approach, with predictions of embeddedness and depth fines being much closer to actual values. For the other Clearwater River drainage sites, the cumulative approach produced over estimates of actual embeddedness and abundance of fines. Predictions of summer fish densities were also much too high with this approach.

Egg to Alevin Survival

Egg to alevin survival studies in 1986-87 represented minimum survival values because of the potential for loss of emerging larvae through the unclosed tops of the egg baskets and possible incursion of predators. This precludes use of 1987 results for developing anything more than the relationship between depth fines and the minimum survival that can be expected. This situation was largely rectified in the 1987-88 plantings in the Poverty Flat study sites, though some cases of ruptures in the tops of the baskets occurred. In many cases, complete failure of the artificial egg pocket to produce viable young could be traced to these causes and excluded from analysis.

FISHSED1 generally predicted higher survival than was actually observed (Figure 2), even when there was no larval escapement through open basket tops. There were a few instances of actual survival being higher than predicted, particularly in the 1987-88 plantings, and two of the observed survival figures for the NFRR site were much higher than predicted; survival in these cases exceeded predictions by 39 and 51%, respectively.

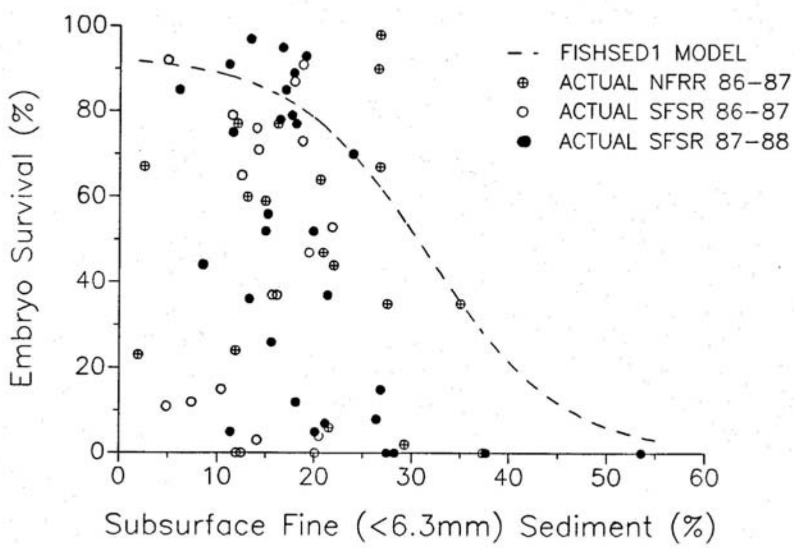

Figure 2.--Egg to alevin survival success as a function of fine sediment (<6.3mm) concentration in artificial egg pockets, Poverty Flat (SFSR) and North Fork Red River study areas, 1986-1987.

In some ways, the actual point scatters from the the field data analysis (Figure 2) were similar to the data scatter in FISHSED1's chinook survival response curve (see Stowell et al. 1983). Specifically, there was potentially high but quite variable survival in the 15 to 30% depth fines range. Considerably more instances of very low embryo survival occurred in the field study than are shown on FISHSED1's survival curve. This was possibly due to inherent differences between field and laboratory conditions, or weaknesses in the design of the field tests (e.g. potential for alevin escape and predation). For the Poverty Flat data, there is no evidence than any embryo survival can be expected when depth fines in the egg pocket exceed about 27%.

It is interesting to note that embryo survival in the NFRR site was apparently higher at higher levels of depth fines than in the Poverty Flat site. These salmon may be better adapted to relatively greater levels of fines, or other environmental factors may have moderated the effects of the greater amounts of depth fines. Red River in the areas sampled is, in fact, a "dirtier" system than the SFSR areas sampled.

Embeddedness and Summer Rearing Capacity

Summer rearing densities predicted by FISHSED1 were clearly many times higher than actual densities for trout, while actual chinook fry densities were higher than predicted. Plotting of our actual trout fry densities against actual (ocular) embeddedness produced a positive relationship, whereas chinook salmon fry densities exhibited the expected negeative relationships (Figure 3).

DISCUSSION

Weaknesses in FISHSED1 belong to two categories: structural weaknesses related to the validity of the laboratory studies used to develop the model and to the ability of the coefficients selected in generating accurate predictions; and philosophical difficulties related to model design. Both of these categories of problems present users with difficulties in relating model inputs (e.g. R1-R4 model outputs) and outputs (e.g. FISHSED1 predictions) to the real-world situations they face. It must be noted that some of the difficulty with respect to the philosophical failings of FISHSED1 are undoubtedly created by inadequacy in the R1-R4 model. Specifically, the R1-R4 model leaves the user of FISHSED1 with no indication of the expected validity of its outputs and without any clue as to how to incorporate the related factors of sediment disposition within the critical reach and overall watershed recovery processes. The fact that FISHSED1 fails to allow for watershed recovery or in-reach sediment storage is probably due to the same failure in the R1-R4 model.

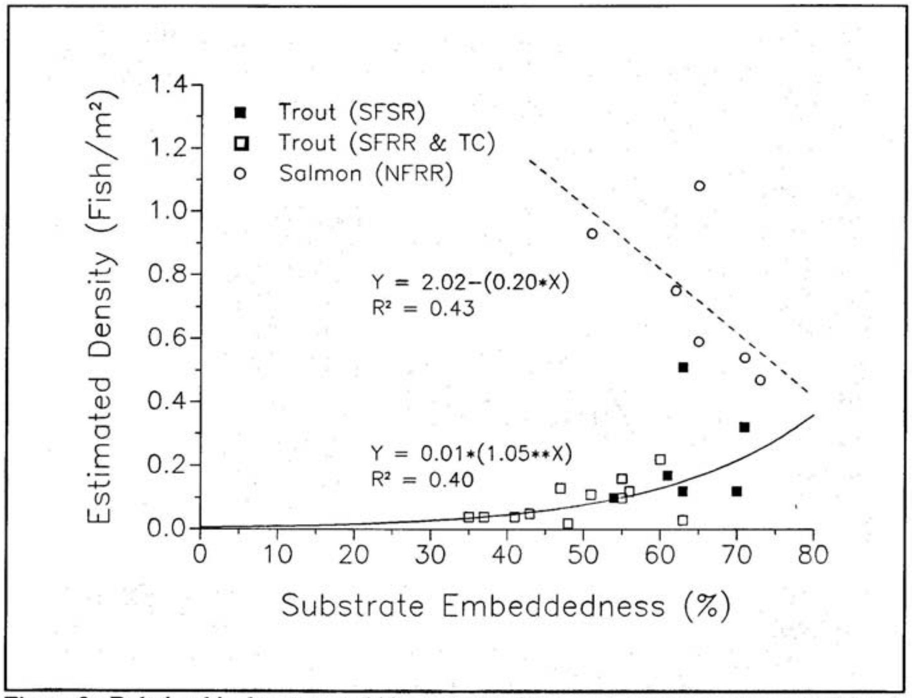

Figure 3.--Relationship bewteen cobble embeddedness and summer trout and chinook salmon fry densities, Salmon and Clearwater River drainages, 1984-1986.

The fundamental philosophical weakness with the FISHSED1 modeling process is an overly simplified view of the world. This weakness is expressed in several ways, but particulary in the way changes in the watershed from year to year are poorly incorporated, and in the way potential ecological interactions are not readily incorporated into use of the model. Such philosophy may be expedient, but it represents bad ecology.

Ecosystem functioning is largely ignored in the straightforward application of FISHSED1. The significance of this fact is that the user of is generally expected to input one data point representing sediment delivery over "natural" levels (which may themselves be unknown) to derive predictions of changed habitat conditions and how this altered state will impact local fish populations. In this process, only influences from actual or potential changes in delopment activity are used to generate a prediction of increased sediment delivery; the influence of climate is largely ignored, the cyclic nature of anadromous fish populations is ignored, random fluctuations in resident fish populations are ignored, the influences of competition are ignored, the influence of the specific local riverine-riparian habitat, and the status of other important ecosystem variables that may influence populations are ignored.

The clearest example of the lack of an "ecological perspective" is the search for a single limiting factor controlling fish populations instead of incorporating the effects of ecological interactions in modeling fisheries responses to management activities is reflected in our embryo survival results. Embryo survival in the SFSR was highly variable over the range of zero to about 20% depth fines, and ranged from zero to 100% survival. Fitting a precise mathematical model to this sort of data seems unrealistic, as reflected in the inability of FISHSED1 (or models we are trying to develop for FISHSED2 for users who may need such definitive predictions) to predict actual survival. This is a structural problem that have attempted to remedy by modelling "survival zones" rather than percent survival (Figure 4).

The first zone, from 0 to about 20% fines we have termed this region the "Good Zone" because average survival was quite high, but the high variability suggested that other environmental factors were generally more important determinants of survival than depth fines. From about 20 to 27% depth fines, survival remained variable but appeared to generally be declining. We have termed this region of the curve the "Marginal Zone" because survival was only moderate, suggesting that depth fines were becoming increasingly important as determinants of survival. We have termed the region above 27% depth fines the "Dead Zone" because it appears that depth fines become an absolutely limiting factor in this range, and little, if any, survival can be expected. Consequently, functional relationships used to model survival in FISHSED2 ignore this region, attempting to predict survival only in the region where there is some real liklihood of survival. This may not actually be a weakness with the model *per se*, but perhaps more with the way that users are left to determine whether the simple single limiting factor (e.g., sediment) approach is suitable, given the current emphasis on sedimentation problems.

Since there may be a need by users to estimate survival potential with a more definitive model than the survival zone approach, mathematical models for survival as a function of large (<6.3mm) and small (<0.8mm) fines will be included in FISHSED2. One potential model is included in Figure 6, but the formulation of these models is still underway.

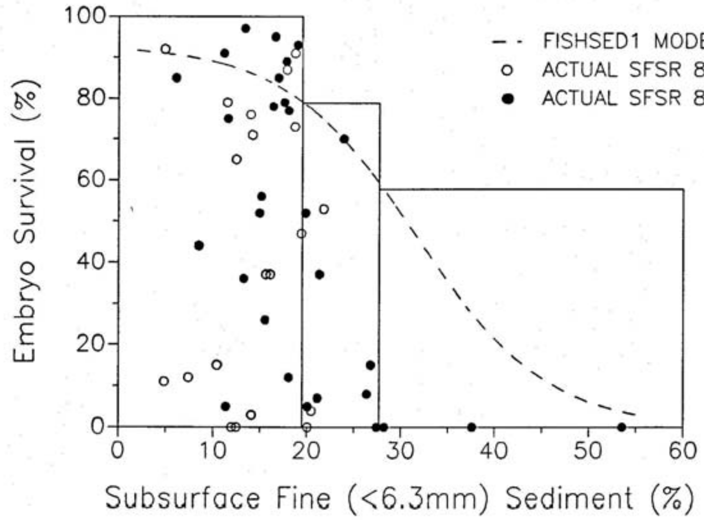

Figure 4.--Propsed "Survival Zone" structure for prediction of chinook embryo survival from fine sediment (<6.3mm) concentrations, South Fork Salmon River. From left to right: "Good Zone", 67% survival; "Marginal Zone", 27% survival; "Dead Zone".

Similarly, FISHSED1 attempted to predict the carrying capacities of salmonids, as reflected by actual densities, without regard to the nature of the local fish community. Thus, FISHSED1 would predict the same density of steelhead trout in a stream reach regardless of the presence or absence of potential salmonid competitors or other species. Structurally, we found no negative relationship between embeddedness and trout fry densities, and will incorporate the positive relationship of Figure 3 in FISHSED2; the chinook salmon relationship used by FISHSED1 was adequate, and FISHSED2 will include it and the relationship in Figure 3 as modelling options.

24

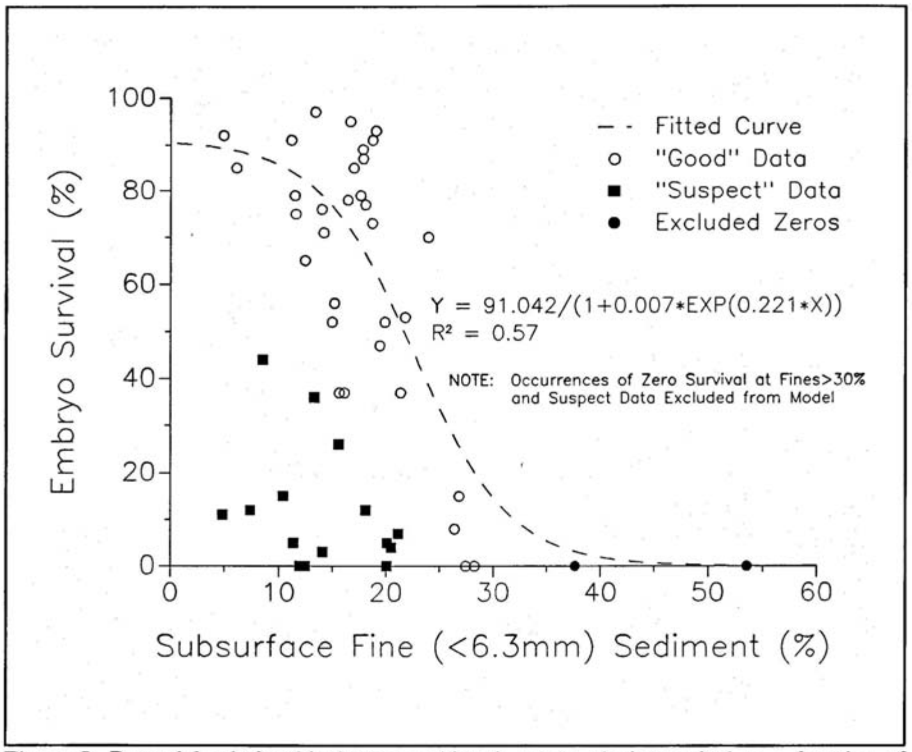

Figure 5.--Potential relationship between chinook egg to alevin survival as a function of fine sediment (<6.3mm) concentration, South Fork Salmon River.

Although FISHSED1 has been shown to be imperfect, it represented, until now, the best available method for estimating the potential sedimentation effects on fish populations and habitats from proposed forest management alternatives. In order to avoid "throwing out the baby with the bathwater," it has seemed prudent to retain the conceptual approach used in FISHSED1 and attempt to refine the functional relationships. Our offering, FISHSED2, is not intended to be an end product, it is more likely another transitional product, subject to further revision as our understanding of the relationships among forest management activities, sediment, and fish expands.

LITERATURE CITED

CHAPMAN, D.W., and K.P. McLeod. 1987. Development of criteria for fine sediment in the Northern Rockies ecoregion. Final Report. U.S. Environmental Protection Agency, EPA 910/9-87-162, Seattle, WA. 279p.

CLINE, R., G. Cole, W. Megahan, R. Patten, and J. Potyondy. 1981. Guide for predicting sediment yields from forested watersheds. U.S. Department of Agriculture, Forest Service, Northern and Intermountain Regions, Missoula, MT, and Ogden, UT. 41p.

COLLOTZI, A.W. 1974. A systematic approach to the stratification of the valley bottom and the relationship to land use planning. p.484-497 IN: Instream Flow Needs, Volume 1. American Fisheries Society, Bethesda, MD.

ESPINOZA, F.A., Jr., R. Patten, and D. Wilson. 1981. Aquatic-ecological land units for the Clearwater National Forest. Unpublished report. U.S. Department of Agriculture, Forest Service, Clearwater National Forest, Orofino, ID.

EVEREST, F.H., R.L. Beschta, J.C. Scrivener, K.V. Koski, J.R. Sedell, and C.J. Cederholm. 1987. Fine sediment and salmonid production: a paradox. p98-142 IN: E.O. Salo and T.W. Cundy. 1987. Streamside Management: Forestry and Fishery Interactions. College of Forest Resources and Institute of Forest Resources, University of Washington, Seattle, WA.

FENNEMAN, N.M. 1931. Physiography of western United States. McGraw Hill Book Company, Inc., New York, NY. 534p.

HILLMAN, T.W., J.S. Griffith, and W.S. Platts. 1987. Summer and winter habitat selection by juvenile chinook salmon in a highly sedimented Idaho stream. Transactions of the American Fisheries Society. 116:185-195.

MCCUDDIN, M.E. Survival of salmon and trout embryos and fry in gravel-sand mixtures. M.S. Thesis, University of Idaho, Moscow, ID. 30p.

PLATTS, W.S., and M.L. McHenry. 1988. Density and biomass of trout and char in western streams. U.S. Department of Agriculture, Forest Service, General Technical Report INT-241, Intermountain Research Station, Ogden, UT. 17p.

PLATTS, W.S., W.F. Megahan, and G.W. Minshall. 1983. Methods for evaluating stream, riparian, and biotic conditions. U.S. Department of Agriculture, Forest Service, General Technical Report INT-138, Intermountain Forest and Range Experiment Station, Ogden, UT. 70p.

POTYONDY, J.P. 1988. Boise National Forest cobble embeddedness baseline inventory: results and relationship to management activities. U.S. Department of Agriculture, Forest Service, Boise National Forest, Boise, ID. 37p.

ROSGEN, D., and L. Silvey. 1980. Channel types for Region 2. Unpublished report, U.S. Department of Agriculture, Forest Service, Ft. Collins, CO.

STOWELL, R., A. Espinoza, T.C. Bjornn, W.S. Platts, D.C. Burns, and J.S. Irving. 1983. Guide for predicting salmonid response to sediment yields in Idaho Batholith watersheds. U.S. Department of Agriculture, Forest Service, Northern and Intermountain Regions, Missoula, MT, and Ogden, UT. 95p.

TAPPEL, P.D., and T.C. Bjornn. 1980. Effects of sediment composition on salmonid egg survival. M.S. Thesis. University of Idaho, Moscow, ID. 20p.

TORQUEMADA, R.J., and W.S. Platts. 1988. A comparison of sediment monitoring techniques of potential use in sediment/fish population relationships. Unpublished report submitted to Idaho Department of Fish and Game, Boise, ID.