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Discussion

Environmental impact assessments should include rigorous scientific peer review

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ABSTRACT

Twenty USA states or jurisdictions and 125 nations have modeled national environmental policies after the National Environmental Policy Act. That act mandates that federal agencies initiate environmental impact statements (EISs) when substantive environmental or human health consequences are likely because of an agency action related to proposed development projects. The science used to inform the EIS process, however, does not require independent scientific peer review (ISPR) in the USA or most other nations. But ISPR is needed for governments to accurately inform the EIS decision-making and public reporting processes. Instead, science is routinely manipulated during EIS reviews to generate expedient project outcomes with substantially negative ecological, political, and long-term economic consequences. We provide four examples of EISs that lack ISPR, as well as four examples where reviews by independent scientists were helpful to improve agency decisions. We also recommend that independent scientists (no affiliation with the project proponents or agencies overseeing projects) be used to help assess potential environmental and socio-economic impacts, as well as offer appropriate risk assessments, study designs, and monitoring timeframes. We conclude that nations should convene formal reviews using independent scientists as a form of peer review in the EIS process.

1. Introduction

1.1. *Short-changing the environmental review process is damaging to the environment*

The bi-partisan USA Congress and President Richard Nixon enacted The National Environmental Policy Act (NEPA) in 1970, a global model in environmental assessment policies (USG, 2023). NEPA grew out of increased public awareness for the environment that culminated in the 1960s with its main purpose to help decision-makers estimate the true costs of proposed projects and thereby protect the human environment from major ecosystem impairment at taxpayer expenses (CEQ, 2021). NEPA required two actions. (1) It called for a Council for *Environmental* Quality (CEQ) within the office of the President. (2) It mandated that federal agencies initiate environmental impact statements (EISs) when substantive (i.e., unmitigable) environmental consequences are likely because of an agency action or proposed development project (USEPA, 2023a; USG, 2023). Under an EIS, federal agencies are required to systematically assess the environmental impacts of their proposed actions and consider alternative ways of accomplishing project proposals that are less damaging to the environment. Currently, 20 USA States or jurisdictions and 125 nations have enacted environmental policies modeled after NEPA (Eccleston, 2008; USEPA, 2023a).

Because of the vast numbers of potentially damaging actions, federal agencies routinely use Environmental Assessments (EAs; Eccleston, 2008). Relative to an EIS, an EA is a shorter public document providing

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documentation including analyses for determining whether a federal agency should indicate no significant environmental impact or prepare an EIS. EAs in the USA are much less comprehensive than those required under Canadian law, which resemble EISs and are focused on projects likely to cause substantial adverse environmental impacts, such as major new mines, marine terminals, highways, or waterways (CEAA, 2023). Likewise, they differ from the more rigorous EIAs required by the European Union for new major power stations, transportation projects, dams, or waste disposal facilities (EU, 2014).

1.2. A fundamental flaw in environmental impact statements (EIS)

We are a group of scientists involved for several decades in reviewing and preparing EISs for proposed projects affecting the environment (e.g., oil & gas, forestry, and mining) and planning for conservation and environmental protection in the USA and abroad. Our observations indicate persistent failure of the EIS process to accurately assess and predict likely harmful impacts to the human and natural environment, especially those involved in evaluating large areas and long timelines.

Many issues present in EIS production and review have been described over the last three decades in the USA and in similar processes globally (Eccleston, 2008). The baseline science, data collection, data handling, and data analysis are often low quality and rushed (Fairweather, 1994; Treweek, 1996; Thompson et al., 1997; Benkendorff, 1999; Ayles et al., 2004; Chang et al., 2013). Risk-assessment models are often poorly justified (Stern, 2013; Sheaves et al., 2016), then feed into impact determinations without identifying model assumptions and levels of uncertainties (Ortolano and Shepherd, 1995; Adelman, 2004; Duncan, 2008; Lees et al., 2016). Consequently, there is a long history of EIS predictions that have proven wrong after project completion. For example, the impacts of mining on water quality are well documented (e.g., Woody et al., 2010; Daniel et al., 2014; Hughes et al., 2016; Salvador et al., 2020), as is the long history of failures to accurately predict those impacts through the EIS process (Kuipers et al., 2006; Woody et al., 2010; CEC, 2022; see Section 2).

We provide our reasoning and examples herein for why independent scientific peer review (ISPR) is needed to effectively weigh environmental (environment, social, and economic) risks. An ISPR facilitates complying with statutes like NEPA regarding the likely impacts of actions that substantially affect local, regional, or global systems upon which humans depend. The EIS process is founded on the idea that an independent scientific assessment of proposed project alternatives and their potential impacts better inform final decisions. By independent, we mean environmental experts who are not affiliated financially with project outcomes.

1.3. The cover of "best available science"

As presently conducted, most EISs are flawed because agencies routinely use what they consider as the "best available science," which is often inadequate for at least five reasons. (1) The science is often not the best or current. (2) The project proponent ignores or downplays the internationally accepted precautionary principle and the burden of proof standard of no or least harm on the part of the project proponent (DellaSala et al., 2022). (3) Type-I over Type-II errors are emphasized (McGarvey and Marshall, 2005; Joly et al., 2010). (4) Data with insufficient statistical power are used. (5) Unreasonable *p* values are set (McGarvey, 2007; Wasserstein and Lazar, 2016). Also, courts routinely defer to agencies when contradictory evidence is presented in project appeals filed by environmental non-governmental organizations, including when those are backed by legal declarations from well-published independent scientists. Project assessments that focus narrowly on a single stressor or class of stressors, such as water quality (Rau, 2017; Hill et al., 2023), rather than the numerous environmental dimensions that support multiple components of healthy aquatic biota, are a special concern (Karr and Dudley, 1981; Karr, 1991; Karr et al., 2022). Likewise, assessments that focus on an immediate site, versus an entire drainage basin or airshed through time, are flawed from the start (FEMAT, 1993; Henjum et al., 1994; USEPA, 2023b).

The claim of "best available science" is routinely manipulated to generate pre-desired project outcomes often by "cherry picking" the science that supports a preconceived or desired outcome (McGarvey, 2007; Collard and Dempsey, 2022; Baker et al., 2023; Collard et al., 2023). Proposed projects frequently do not require "the federal government to use all practicable means to create and maintain conditions under which man and nature can exist in productive harmony" (USEPA, 2023a). Instead, agencies increasingly tend to sidestep more comprehensive EISs for overly simplistic EAs and by proposing projects that seek categorical exclusions for multiple projects deemed inconsequential when they truly may not be. We recognize that decision-making is ultimately influenced by political and economic realities (Dillon et al., 2018; Eccleston, 2008). However, the environment and the public interest are not well-served when science is manipulated to yield anticipated outcomes unsupported by ISPR. In other words, socio-economic and political decisions should be clearly separated from objective science and the need for evidence-based decisions (Hughes et al., 2021, 2023). Our criticism is not new (Lessing and Smosna, 1975; Schindler, 1976; Hilborn and Walters, 1981; Bella, 1987; Fairweather, 1989; Buckley, 1989, 1991; Peterson, 1993); however, the negative consequences are now magnified by the biodiversity and climate crises. In addition, federal agencies in the USA (and likely in other nations) often avoid more comprehensive analyses to bypass discoveries that are provided by more detailed environmental and economic analyses. Proponent agencies seek exemptions (e.g., categorical exclusions) under NEPA, asserting without evidence that projects are inconsequential when they are not so environmentally nor economically. A robust ISPR would limit flawed EIS and EA outcomes.

2. Four examples of flawed EISs

2.1. Trans-Alaska Pipeline System (TAPS)

An early example of an EIS is that developed for the Trans-Alaska Pipeline System (TAPS).

Large petroleum deposits were discovered in northern Alaska in 1968. After further exploration, a consortium of oil companies applied for a federal pipeline permit in 1969, planning to bury the pipeline and ship heated oil through it. But Native Alaskans, whose lands the pipeline would pass, sued in 1970. That issue was resolved in 1971 via the Alaska Native Claims Settlement Act at a price of \$962 million plus 149 million acres of federal land returned to Native Alaskan entities. But then several environmental groups sued, stating that the pipeline would violate the Mineral Leasing Act. They also warned that the companies failed to consider alternative routes and such potential environmental impacts as oil spills, permafrost melting, earthquakes, erosion at >500 road-stream crossings, pipeline expansion and contraction, and fish and wildlife habitat losses. In response, the Department of Interior published an EIS in 1972. The 1300-km pipeline was completed, and oil began flowing in 1977. On March 24, 1989, the *Exxon Valdez* ran aground and spilled >4000 m^3 of oil into Prince William Sound (Fig. 1), killing billions of fish and thousands of birds and mammals (Piatt and Ford, 1996). The oil companies and the Department of Interior had failed to consider the risks of shipping oil in a single-hull tanker where free ice occurs and oil recovery is extremely difficult. The spill cost Exxon at least 1 billion USD, and the Prince William Sound ecosystem has yet to fully recover, at least partly because oil seeped into cobbles and PAHs (polycyclic aromatic hydrocarbons) in the oil had persistent embryotoxic and trophic cascade effects (Peterson et al., 2003; Incardona et al., 2015; Barron et al., 2020). Failing to embrace an ISPR and adequately consider and mitigate all the risks of moving oil across the seascape remains an economically and environmentally costly decision.

Fig. 1. The *Exxon Valdez* spilling oil after running aground in Prince William Sound (from: RGB Ventures/SuperStock/Alamy Stock Photo).

2.2. Pebble Mine (Bristol Bay, Alaska) draft EIS

The USA Army Corp of Engineer's Pebble Mine draft EIS (ACOE, 2019) failed to consider ISPR and displays more of the inadequacies in NEPA analysis and implementation by USA federal agencies. Located on State land in Alaska's Bristol Bay watershed, the mineralized region is 13.5 km^2 by 600–1200-m deep (Woody, 2018). The initially proposed Pebble Mine would have required a 760-m deep by 3.7-km wide open-pit with a 166-m high tailings dam (Chambers et al., 2012). During mine operation, the treatment and discharge of 54 billion L of water annually would be required (ACOE, 2020). Underground mining of additional ore, not analyzed in the EIS, would almost certainly follow the open pit mining phase, and would require additional tailings facilities.

The pristine rivers draining the mine claim are essential to salmon and flow into Bristol Bay, home to the world's largest wild sockeye salmon (*Oncorhynchus nerka*) fishery (Woody, 2018). Its seafood industry

Fig. 2. Example of a catastrophic tailings dam failure. Tailings from the Polley Mine spill deposited over Hazeltine Creek (https://watershedsentinel.ca/articles/mount-polley-mine-is-still-pumping-waste-into-quesnel-lake/). Before the spill, the creek was 5-m wide, with a cobble and gravel bed and forested riparian zone (Byrne et al., 2018).

employs thousands of people and generates millions of USD in sales annually. Indigenous peoples have subsisted on salmon from the rivers for millennia; salmon comprise 60–80% of their traditional harvest and have averaged >100,000 salmon annually (Woody et al., 2010). Metal mines have a long history of point-source water pollution and tailings pond failures (Kuipers et al., 2006; Woody et al., 2010; Bowker and Chambers, 2017; CEC, 2023), some that have been catastrophic (e.g., Escobar, 2015; Virgilio et al., 2020). The USA Environmental Protection Agency's peer-reviewed Bristol Bay Watershed Assessment found that the mine could have "unacceptable adverse effects on fishery areas" (USEPA, 2014).

Nonetheless, in its EIS the ACOE, 2020 did not even consider a catastrophic tailings dam failure, despite recent calamities in Brazil and at the Mount Polley facility in nearby British Columbia (Fig. 2). Analysis of catastrophic dam failures is necessary to protect workers and the public, to ensure that appropriate warning systems and evacuation plans are developed, and to avoid building facilities in areas that could be inundated. Finally, the ACOE concluded that mining would not have "a measurable effect" (ACOE, 2020) on Bristol Bay fisheries.

The USEPA (2019) had called attention to a litany of the ACOE's draft Pebble Mine EIS problems. Those included that the draft lacked details regarding waste-rock chemical characterization, ground-water modeling, wetland and stream impacts, marine impacts, mine dewatering, tailings dam and water management, ground water seepage, water treatment plant operations, fishery impacts, mine reclamation, environmental monitoring, compensatory mitigation, and risks of tailings facility failures.

In November 2020, the ACOE (2020) reversed its prior draft EIS, which asserted that the mine would have no substantive effects, and denied Pebble Mine a permit, stating that its plan failed to "comply with Clean Water Act guidelines," was "contrary to the public interest," and offered an "insufficient amount of compensatory mitigation." Following appeals, the USEPA (2023b) also denied Pebble Mine a permit to use the Bristol Bay watershed for disposing of mine dredged or fill material under Clean Water Act Section 404(c). In a 280-page document with 340 peer-reviewed references, citing science-based decisions, the USEPA deemed the mines would cause unacceptable harm to salmon. Thus, despite an inadequate draft EIS and after over 20 y of prospecting, litigation, research, hearings, and subsequent ISPR, the Pebble Mine was effectively defeated. But that should have been obvious even before the draft EIS to any trained ecologist, fishery biologist, or environmental scientist. It was simply too vulnerable a place for a mega-mine.

2.3. Alaska's general mine EISs

Hardrock mines use and generate large volumes of hazardous and toxic materials that have substantial environmental and public health risk when spilled. These spills include processing chemicals (e.g., cyanide solution), ore concentrates (e.g., heavy metals), fuels, or mine tailings. Mining EISs rarely quantitatively address spill risks, and generally only consider spills related to transportation. The Alaska Department of Environmental Conservation (ADEC) maintains a thorough and publicly accessible database of mine permitting documents and reported spills. For the five major hardrock Alaska mines (Pogo, Greens Creek, Kensington, Fort Knox/True North, and Red Dog), the transportation spills model used by USEPA (2014) would have predicted a total of 4.3 truck accidents with hazardous material spills (Fig. 3A) if such analyses had been shown for all five mines. That number of spills (N) was based on the predicted number of kilometers traveled (T) for all five mines from their beginning production dates through 2020 and a spill rate per kilometer (R) using $N = RT$. Lubetkin (2022) compared the spill predictions in permitting documents versus spill records for 1995–2020 from ADEC (2021) records and showed that there were 1004 total transportation-related spills at all five mines, resulting in aggregate totals of 127 m^3 and 803,347 kg of hazardous materials spilled (Fig. 3A; Lubetkin, 2022).

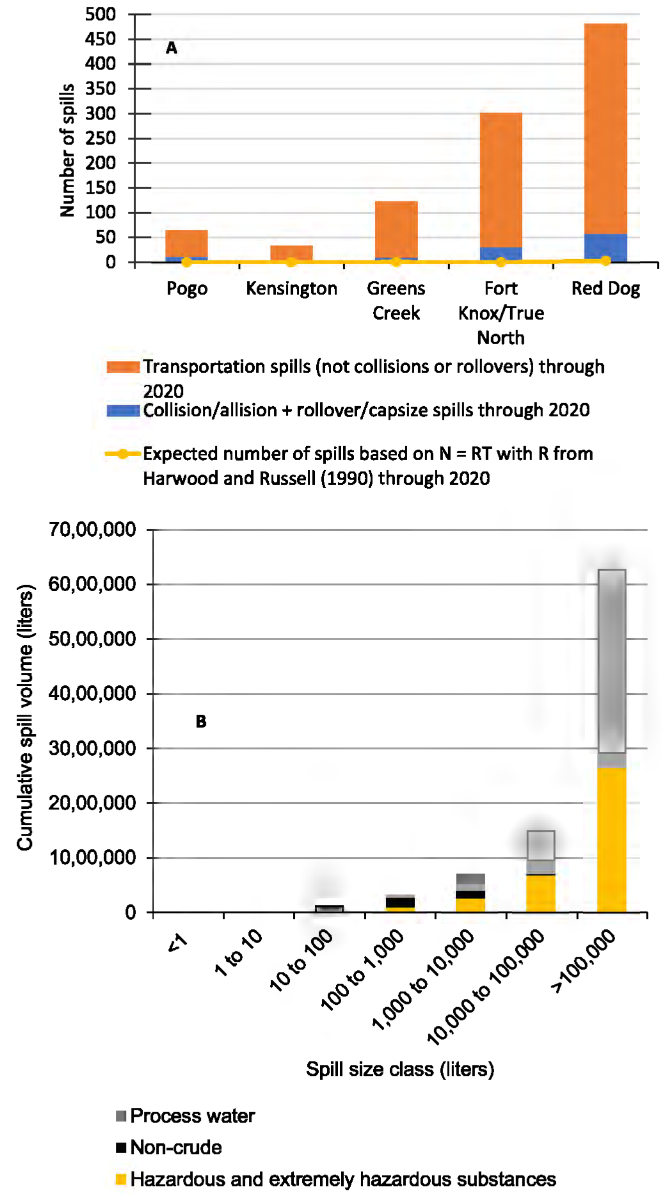

Fig. 3. (A) Number of transportation spills at five Alaskan mines (Pogo, Greens Creek, Kensington, Fort Knox/True North, and Red Dog); and (**B**) spill volumes by substance class from all sources and causes at those five mines from their beginning production dates through 2020 (Lubetkin, 2022; Harwood and Russell, 1990).

However, transportation spills were a small portion of the total number of spills reported at the five mines. The ADEC database for the five hardrock mines documented more than 8150 total spill incidents, releasing >8,934,000 L and >875,433 kg of hazardous materials since 1995 (Fig. 3B; Lubetkin, 2022). Many of the substances that were listed in the ADEC spills database were not mentioned in the EISs as part of reagent lists, fuels, or tailings that could be released (Lubetkin, 2022).

Within the EISs, spills of individual substances were described as low-probability events, and the aggregate, cumulative risks and impacts of all the hazardous material spills from all sources and causes were not addressed. Both the EISs and EAs lacked explicit, complete, and quantitative reagents lists, as well as specifications of other chemicals used for blasting, water treatment, and spill mitigation, that would be considered as hazardous materials being transported to or from the mine or used on-site. Based on our general concerns, we assert that mining EISs should be improved in seven ways.

- Include explicit, complete, and quantitative hazardous material lists for substances transported to or from the mine or used on-site.
- Provide complete descriptions of the transportation methods, load sizes, and transportation frequencies for the hazardous materials.
- Include newly built mine roads as well as the origins and destinations of the hazardous materials in the transportation corridor modeling.
- Ensure realistic quantitative transportation spill-risk estimates for the aggregated total of trips and the whole mine operation's cumulative hazardous materials spill-risks based on updated available evidence.
- Provide detailed transportation spill risk models, with updated risk-rates and location-specific descriptions of the transportation corridor.
- Model the multiple transportation-related releases, as well as likely accidents.
- Enumerate the numbers of expected spills, even if those estimates are minimum values, because there are insufficient data to model all potential spill causes (i.e., apply the precautionary principle).

Spill risks were the only aspect considered in the EISs and EAs of the five mines (Fig. 3A) examined by Lubetkin (2022), but they exemplify how decision-makers and community members receive insufficient representations of the environmental consequences of approving large mines. The ISPR by Lubetkin (2022) showed that the spill-risk predictions in the EISs and EAs were incomplete, inaccurate, or nonexistent. Current risk-assessments in EISs for Alaskan mines do not measure up to the main objectives of an informed EIS, which are: (1) estimate potential consequences of project impacts, and (2) inform stakeholders and decision makers how to mitigate those consequences.

2.4. *Characterizing the probability of catastrophic discharge events on the outer continental shelf*

Even when risks are calculated using defensible models, the results may not be put into context such that lay people and decision makers will understand their implications, especially if those persons are unfamiliar with statistical terminology. For example, consider the Bureau of Ocean Energy Management's 2019–2024 National Outer Continental Shelf Oil and Gas Leasing Draft Proposed Program (BOEM, 2018) treatment of the probability of catastrophic discharge events (CDEs). The estimated return periods ranged from 39 y for spills >150,000 barrels (bbl; 23,848 m^3) to 770 y for spills >10,000,000 bbl (1,589,873 m^3; Tab le 7.4 in BOEM, 2018).

Like flood-risk estimates for rivers (Gordon et al., 1992), spills are stochastic events that do not occur with regularity (Friel et al., 1993). An event such as the 2010 *Deepwater Horizon* disaster that released 4.9 million bbl (779,038 m^3) may be estimated to have a return period of >400 y, but that does not mean it can then be safely assumed that the next such event will not occur until after the year 2400. Instead, the return period for a spill of a given volume can be used to find the probability of at least one spill in each time period. For example, if the return period for a specific spill volume is 165 y, then that spill size (or larger) "is most likely to occur once in 165 y, and every year has a chance of occurring of 0.6% (=1/165)" (Ji et al., 2021). Having an estimate of the probability of occurrence in a single year facilitates calculating the probability of at least one occurrence within a specified number of years using the binomial distribution, assuming that the events are independent and identically distributed across all years (Lubetkin, unpublished data).

For example, if the return period for a spill is 165 y, then the probability of having a spill in one year is 1/165 as stated above and the probability of having zero spills in one year is 1–1/165. In that case, the probability of zero spills in two years is $(1–1/165)^2$ and the probability of having at least one spill in two years is 1 – $(1–1/165)^2$. If the return period for a >1,000,000 bbl (158,987 m^3) spill is 165 years (Ji et al., 2014a,b) and the exposure is 30 years, then the probability of at least one spill >1,000,000 bbl = 1 – $(1–1/165)^{30}$ = 16.7%. Analogous computations can be used to find the probability of other spill volumes with their return periods for different amounts of exposure. Thus, using the return

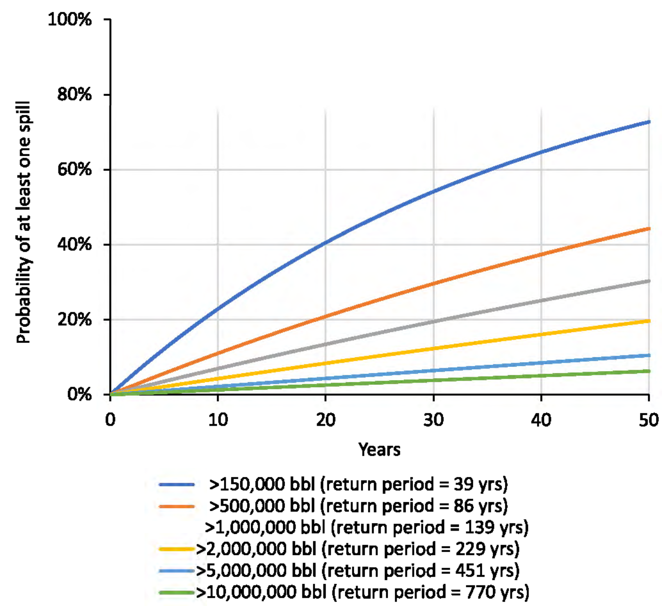

Fig. 4. Outer Continental Shelf spill probabilities for various return periods for spills >150,000 bbl to >10,000,000 bbl (data from BOEM, 2018). One barrel = 0.16 m^3.

periods given in BOEM (2018) for several spill sizes, probabilities can be calculated for various years of production (Fig. 4). BOEM (2018) only presented the return periods and described CDEs as "statistically unexpected" events that "would be considered well outside the normal range of probability" for the 2019–2024 proposed program. But BOEM (2018) showed no calculations for the probability of a CDE for either the 2019–2024 proposed program nor for the outer continental shelf's extant production. A quantitative ISPR (Lubetkin, unpublished data) came to much different, and more realistic, conclusions (Fig. 4).

3. Independent scientific peer review: case study examples

3.1. *Forest ecosystem management assessment team (FEMAT) and the Northwest Forest Plan (NWFP)*

In the 1990s, regional protests over old-growth logging, federal timber-sale injunctions, and the threatened northern spotted owl (*Strix occidentalis caurina*) led to a court injunction on logging by USA District Court Judge William Dwyer. He found that federal agencies were not complying with the population viability standard of the National Forest Management Act and directed the USA Forest Service and the Bureau of Land Management to adopt the landmark Northwest Forest Plan (DellaSala et al., 2015). The plan was kicked off at a Northwest Forest summit in 1993 attended by President Bill Clinton, Vice-President Al Gore, and several cabinet-level officials. Its principal objective, as stated by President Clinton, was to produce a plan that would be "insofar as we are wise enough to know, scientifically sound, ecologically credible, and legally responsible" (FEMAT, 1993).

To ensure that the plan had the scientific foundations needed to comply with the injunction and the President's wishes, six federal agencies involved in the region's forests and wildlife management convened a scientific panel. The panel was charged with developing the justifications and alternatives for ecosystem management, biodiversity conservation, and timber supply (FEMAT, 1993). A key component of the resulting forest plan was the establishment of unlogged forest buffers along streams using buffer widths based on habitat factors needed to support salmonids (FEMAT, 1993; Olson et al., 2007, Fig. 5). The buffer widths were defined as two potential tree heights (100 m) on both sides of streams supporting fish and one tree height (50 m) on both sides of streams lacking fish. Since the development of this standard, state

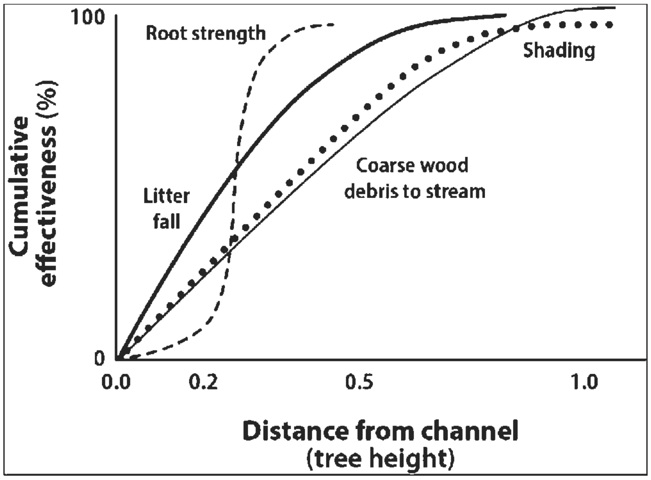

Fig. 5. Cumulative effectiveness of habitat factors contributing to instream habitat conditions for supporting salmonids (FEMAT, 1993).

agencies in the region have established similar (but narrower) stream-side buffers to protect the physical, chemical, and biological conditions of forest streams (Knutson and Naef, 1997; Quinn et al., 2020). Similar buffers have been recommended for agricultural streams (Sweeney and Newbold, 2014; Hughes and Vadas, 2021) and are required for Brazilian streams (Brasil, 2012).

The Northwest Forest Plan is a 100-y plan to recover old forest ecosystems that support imperiled species as well as the transition of rural communities from logging older forests to less unstable economies. As such it is a legal compromise—based partly on ISPR–amongst many competing interests, and it continues to regulate forest management in the USA Pacific Northwest to a degree. The Plan slowed logging of old trees on federal forestlands and led to a quantitative forest monitoring and assessment program (AREMP, 2023; Dunham et al., 2023). However, the spotted owl has continued to decline because of increased competition with non-native barred owls (*Strix varia*; Rockweit et al., 2023) and the climate crisis. Threatened salmon species have not recovered in the Plan area, being limited by the climate crisis, stream fragmentation, and habitat degradation, mainly on nonfederal lands (Gaines et al., 2022). However, without the ISPR that led to the Plan and improved forest management, those additional stressors would have had even more negative impacts on forest ecosystems.

Whereas the NWFP has withstood multiple attempts to weaken its conservation framework for over three decades, it is scheduled for a major revision in 2024. Rather than adopting a FEMAT approach, the US Forest Service convened a stakeholder team consisting of interest groups from timber, conservation, development, tribes, and some scientists. For the most part, that team has ignored the underlining foundation of the original plan such as reserve design, connectivity, spotted owl habitat, and salmonid habitat. In addition, it is ignoring more recent concerns over carbon accounting when assessing older forest stands and fluxes from logging and fires. Thus, the science approach of FEMAT was replaced by an untested stakeholder-driven process versus the original Plan's emphasis on species and old forest viability.

3.2. Eastside Forest Scientific Society Panel

In 1992, a FEMAT-like process was lacking for managing the 6 million ha in 10 national forests east of the Cascades in Oregon and Washington. That shortcoming stimulated a bipartisan group of seven Congressional members to approach the American Fisheries Society, American Ornithologists Union, The Ecological Society of America, Society for Conservation Biology, and The Wildlife Society to form the Eastside Forest Scientific Society Panel (EFSSP). The EFSSP had two mandates. (1) Identify areas where logging could compromise long-term ecological viability of forest, fisheries, and associated values. (2) Recommend management guidelines to protect those critical areas in the interim while a long-term conservation plan would be developed.

Based on 50 map layers of environmental conditions, the EFSSP provided detailed maps and tabulations of natural resource conditions on six major topics for each national forest (Henjum et al., 1994). They first identified areas of existing old-growth forest, located watersheds critical for fisheries, and mapped roads, streams, and roadless areas by size. Next, they reviewed existing knowledge on the status of fish and terrestrial species likely to be altered by forest management and assessed the status of habitats needed by species of concern.

The EFSSP estimated that old-growth forests covered <15% of their original area in lower elevation forests and that continued logging in unprotected areas would reduce old-growth of all forest types to <10% of forest area in the region, thereby challenging their sustainability. Fisheries and riparian areas were generally in poor condition and the status of numerous terrestrial species was of concern. EFSSP recommended 11 measures to ensure "interim" protection of species and ecosystems as part of a long-term planning process for those national forests.

- Allow no logging of late-successional (mature) and old-growth stands (LS/OG).
- Cut no trees older than 150 years or with ≥20" (50 cm) diameter at breast height (dbh).
- Allow no logging of ponderosa pine stands.
- Allow no logging, road building, or mining in designated aquatic diversity areas.
- Allow no new roads or logging in roadless regions >1000 acres (400 ha).
- Establish protected corridors along streams, rivers, lakes, and wetlands.
- Allow no logging or mining in fragile, erosion-prone areas unless ISPR conclusively demonstrates that it will not degrade soils, release sediment to streams, or slow forest regeneration.
- Allow no livestock grazing in riparian areas unless ISPR conclusively demonstrates that it will not damage those areas.
- Establish a panel with broad expertise to develop long-term management guidelines to aid forest capacity to resist drought, crown fires, and catastrophic disease outbreaks.
- Establish a panel to develop a coordinated strategy for restoring the health and integrity of eastside landscape ecosystems and the processes that they depend on.
- Establish a comprehensive quantitative biomonitoring and bioassessment program.

The ISPR was successful in initially producing an ecological monitoring program (PIBOMP, 2023; Henderson et al., 2005) and interim protections for large (>50 cm) trees that stood for nearly two decades. However, the Trump administration rescinded the rule in favor of logging trees up to 150 years old (up to 76 cm dbh) for purported restoration, fire risk reduction, and resilience purposes. That rule change remains controversial as the U.S. Forest Service only cited science that supported the change (Johnston et al., 2021), whereas others have pointed to flaws in that approach (Mildrexler et al., 2023). Old-growth forests have declined to <3% of pre-settlement amounts and only occur in small, isolated stands (Youngblood, 2001). Yet, logging of large trees recovering from past logging were once again targeted. In 2022, six conservation groups sued the Forest Service for violating several federal listed species and forest laws, including aquatic species impacts. In 2023 and 2024, two different federal judges upheld their suit, deeming the agency actions arbitrary and capricious and ordering an EIS. That new EIS should include ISPR.

3.3. Bristol Bay Watershed Assessment

Rather than focusing solely on the Pebble Mine, USEPA's Bristol Bay

Watershed Assessment (USEPA, 2014), written by 16 federal, state, private, and university scientists, provided a comprehensive evaluation of mining risks to Bristol Bay fishery resources. It addressed both the 25-100-y mine development and operation phase as well as the post-mining phase, during which the site would be monitored, and water and solid-waste treatment would be continued in perpetuity. Mining of other copper deposits in the mining district would require the same monitoring and waste treatment. Assuming collection and effective treatment of all water, and no failures, fishery impacts would result from the loss of 90–151 km of salmonid spawning or rearing habitat. Over the long term, four to 10 streams would lose fish passage and be degraded from road-culvert jams, washouts, and erosion. One to two pipeline failures would likely occur over the mine life, which would release toxic water and sediments, kill fish and invertebrates, and persist for decades before settling into Iliamna Lake. Likely failures of the water and waste collection and treatment systems would result in short-term to perpetual toxic releases. A tailings spill would eliminate 38–48% of the salmon run in the Nushagak River and trout populations would be lost for decades (see Fig. 2). USEPA (2014) formed the scientific foundation for EPA's opposition to the ACOE, 2020 draft EIS that permitted the Pebble Mine and its support of USEPA (2023b) that prohibited it.

Unlike the Pebble Mine EIS (ACOE, 2020), which concluded that there would be no significant impact to fisheries in Bristol Bay resulting from the mine, the Bristol Bay Watershed Assessment found that there would be a significant risk of harm to the fisheries. The Bristol Bay Watershed Assessment underwent several rounds of peer review before the final report was issued. The scientific studies that formed the basis for the Assessment and its conclusions were individually peer reviewed. None of the scientific studies supporting the Pebble Mine EIS were peer reviewed. It is logical to conclude that the more thorough and rigorous application of science in the Bristol Bay Watershed Assessment led to significantly different conclusions about risk to the fisheries than those in the Pebble Mine EIS.

3.4. Klamath river (Oregon, California) EIS

The Klamath River was once a major salmon producer (Gresh et al., 2000). Prior to dam construction beginning in 1918, it produced 650,000–1,000,000 fish. Its upper basin sits in the relatively dry Eastern Cascades Slopes and Foothills Ecoregion of Oregon, where ranching and irrigated agriculture are the major water withdrawals. Further downstream, the Klamath flows through the Klamath Mountains Ecoregion in California, which is dominated by coniferous forest and where logging is a major industry. Four upriver dams blocked salmon passage for over 100 y (Fig. 6); conflicts over water rights resulted in crop losses in 2001 to protect salmon, followed by tens of thousands of salmon deaths in 2002 to protect irrigators. As part of its relicensing agreement in 2007, PacifiCorp, the owner and operator of the four dams, had to install fish passage facilities and make other improvements or remove the dams. PacifiCorp determined that dam removal would be less expensive than continuing to operate the dams and entered into a formal agreement with California, Oregon, the Department of the Interior, the National Marine Fisheries Service, and the Karuk and Yurok Tribes in 2016 to remove the dams. Following lengthy ISPR by engineers, geologists, hydrologists, fishery and wildlife scientists, botanists, water quality biologists, sociologists, and economists, a final EIS was produced by the Federal Energy Regulatory Commission (FERC, 2022). The first dam, Copco 2, was removed in 2023; the remaining three dams are scheduled for removal by 2025 (Davidson, 2023). This will be the largest dam decommissioning and salmon rehabilitation project in the history of the USA and will also begin restoring justice to the Tribes who have depended on salmon for their existence for millennia.

Fig. 6. Iron Gate Dam (53-m high, 226-m long; from Michael Wier, Klamath River Renewal Corporation).

4. Independent peer review benefits

Based on our review of some exemplary EIS projects, we assert that the most important flaw in the EIS process is the failure to require ISPR. Scientific journal manuscripts require ISPR before acceptance for publication. Reports by the USA Environmental Protection Agency's Science Advisory Board and the National Academies of Sciences, Engineering, and Medicine require ISPR. Yet, the science used to inform the EIS process does not require ISPR. But ISPR is needed to accurately inform the EIS decision-making, allow for accountability determination in case of compliance failures, and facilitate the public reporting process. Without formal review by independent scientific experts, attorneys representing NGOs are justified in challenging inappropriate analyses of the predicted project impacts.

We emphasize that best available science depends on critical reviews by independent scientists (Karr and Chu, 1999), economists (ECONorthwest, 2019), and statisticians (Utts, 2021) who are not affiliated financially with project outcomes or agency funding contracts. Objective reviewers with knowledge of the relevant science can assess whether project proponents have properly considered short-versus long-term planning horizons; costs of deferred regulations; potential ecological, social, and economic consequences; cumulative effects; and advances in scientific understanding, among others. Federal agencies often make harmful environmental decisions based on a burden-of-proof standard that underestimates impacts and can result in environmental disasters, such as from mining and fossil fuel extraction (Woody et al., 2010; Hughes et al., 2016; Bowker, 2021). This means that environmental organizations must try to correct those shortcomings via project appeals and litigation (Whittaker and Goldman, 2021; Baker et al., 2023), or taxpayer-funded rehabilitation (USEPA, 2000, 2004). Notably, in a survey of 22 recent EISs, only 27.6% (3672 of 13,291) of the references cited were of articles from peer-reviewed journals (Lubetkin, 2020). ISPR and subsequent evaluation and corrective responses to the peer-review findings help ensure transparency, scientific credibility, and accountability.

When the decision in an EIS is delayed on legal appeal for years because of inadequate, biased science, or inconsistent science and conclusions about the risk to the environment and human health, many of those involved pay a high price for wasted effort and time (Eccleston, 2008). The resulting inefficiencies harm citizens, taxpayers, affected communities, agency personnel, industry, and investors, as well as the environment. Poorly applied science and engineering have systemic consequences, including spectacularly expensive failed plans and project proposals, often with unanticipated or undisclosed harm to the environment and human health (Kuipers et al., 2006; Hughes et al., 2016; Salvador et al., 2020).

This is not surprising. Without ISPR and an impartial evaluation as to whether high quality, objective scientific processes were followed by the EIS proponent, there is little incentive for agencies to articulate uncertainties, risks, and likely impacts, and the process naturally becomes siloed by tunnel vision and driven by project proponent preferences. In general, "good science is not, as some have cynically suggested, merely in the eye of the beholder, nor is it whatever technical information can be cobbled together to support one's predetermined position" (Elliott, 2003, p. 46). This is also certainly true for an EIS or an EA as well as projects that bypass the formal review process via "categorical exclusions" under NEPA when realistically those are anything but nonconsequential (DellaSala et al., 2022).

5. Recommendations

5.1. Project pre-proposals

Often EISs are hampered by the quality of available data. Ideally, project pre-proposals and scoping should incorporate early interaction amongst developers, regulators, the public, and independent scientists to list likely concerns regarding potential ecological and socio-economic impacts, as well as offer appropriate risk assessments, study designs, and monitoring methodologies and timeframes (Eccleston, 2008; Noble, 2020, Fig. 7). Monitoring is often neglected or poorly designed and funded in initial and subsequent EAs and planning, which precludes effective adaptive management (Hughes et al., 2000; Maas-Hebner et al., 2016). In addition, agencies and proponents tend to be married to their ideas and reluctant to change if the first round of planning is less robust than that required in an EIS. Thus, an additional step should focus on the study designs needed to improve the initial- and later-stage science of impact assessment and adaptive management. Particularly important is having adequate sample sizes and statistical power for relevant studies, to minimize Type-II errors that falsely infer no impacts (McGarvey, 2007; Utts, 2021; Hughes et al., 2023), and to ensure that statistical and biological significance are not falsely synonymized (Possingham et al., 2001; Vadas et al., 2022). Conversely, erroneous reporting of statistically significant results resulting from pseudoreplication must be avoided (Perneger and Combescure, 2017; Utts, 2021; Vadas et al., 2022). In many USA and global cases, the impacts and risk assessments of prior, similar projects—and what went wrong with them—are available in the scientific literature and on-line (e.g., Kuipers et al., 2006; Bowker and Chambers, 2017; Bowker, 2021; CEC, 2022).

5.2. Project EIS or EA

If the questions asked in the initial scoping phase are answered satisfactorily, an EIS or EA peer review could take many forms (Eccleston, 2008). Other countries have successfully implemented such reviews. For example, under the Canadian Impact Assessment Act (Government of Canada, 2019), the Minister of Environment and Climate Change may determine that it is in the public interest to refer the assessment to an independent review panel (CEAA, 2023). Such a review panel is a group of independent and impartial experts appointed by the Minister to: (1) conduct the environmental assessment; and (2) make conclusions and recommendations to the Minister. The review panel members must have knowledge or experience relative to the anticipated environmental, social, and economic effects of a project. They must also be objective and free from any apparent financial conflict of interest relative to the project or their own research funding source, such as being under contract for the proposing agency or entity. Some of them should be environmentally concerned scientists, economists, and sociologists. Because of the amount of time required by the peer-reviewers plus the need for independence, the proponent and the oversight agency should fund the reviewers via an independent contractor as several of us have experienced (Table 1).

We encourage the USA Council on Environmental Quality (CEQ) and similar bodies in other nations to adopt the kind of thorough review processes exemplified in Section 3 and Fig. 7, as a component of the EIS process. The extent and complexity of the ISPR should vary with the potential extent and cumulative effects of the project, which are determined during the scoping process (Eccleston, 2008). Public comment on the EIS or EA scoping that is prepared by the cooperating agencies should identify most of the areas requiring peer review of the technical reports used to establish EIS or EA conclusions. Although this adds additional

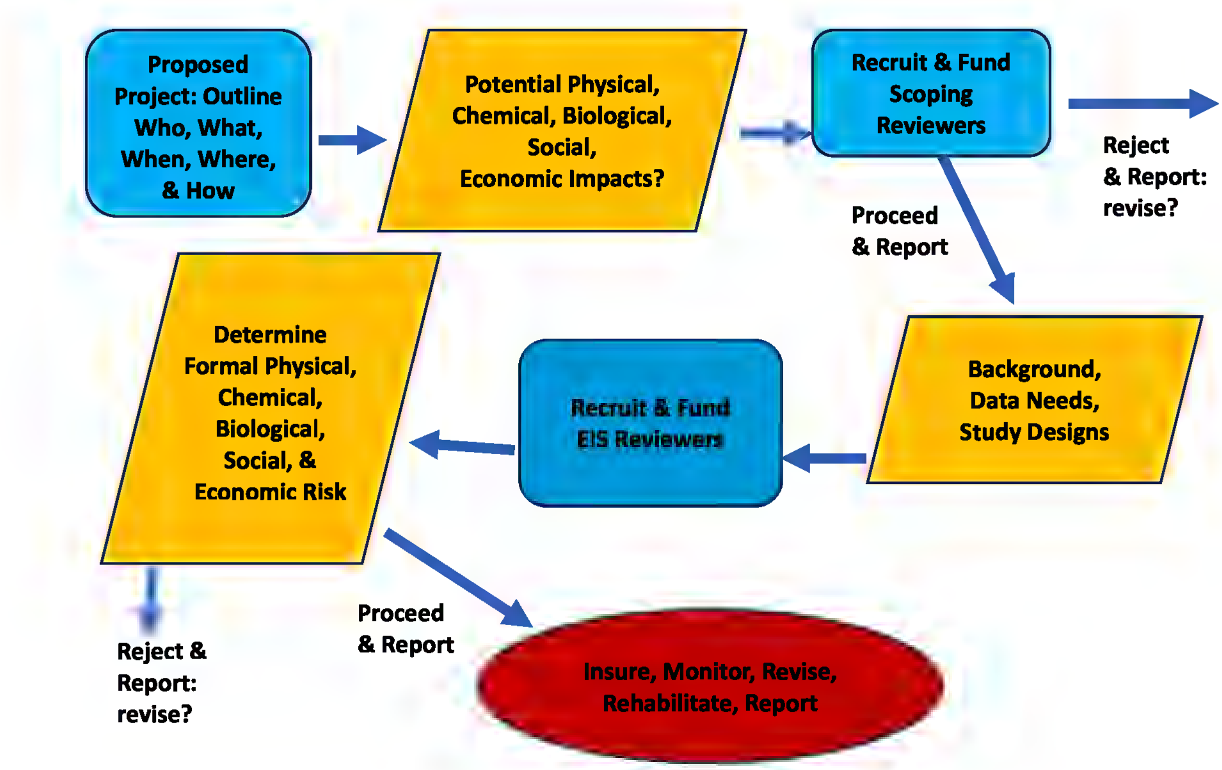

Fig. 7. Recommended steps in EIS independent scientific peer-review process.

Table 1
Examples of scientific peer reviews, their funding sources, and durations.

Name	Funder	Duration	Reference
Aquatic Conservation Strategy	Coast Range Association	2-day meeting 2-month writing	Frissell et al. (2014)
Ecological Conditions in Hydropower Basins	Companhia Energética de Minas Gerais	5-years	Callisto et al. (2014)
European Fish Index	University of Natural Resources & Life Sciences, Vienna	2 1-week meetings	EFI+ CONSORTIUM (2009)
EMAP Indicator Workshop	USEPA	4-day meeting 1-month writing	Hughes (1993)
Gulf of Mexico Catastrophe	Earthjustice	6 months	Lubetkin (2020)
Kissimmee River Restoration	South Florida Water Management District	4-day meeting 2-years writing	Loftin et al. (1988)
Mining & Fossil Fuel Extraction	Not funded	2-years writing	Hughes et al. (2016)
Mining Retrospective	Natl. Parks Cons. Assoc., Tanana Chiefs, Earthworks, Brooks Range	9 months	Lubetkin (2022)
National Wildlife Refuge System	Indiana University Schools of Law and Public & Environmental Affairs; US Fish & Wildlife Service	1-week meeting 2-years writing	Meretsky et al. (2006)
Oil Sands Monitoring	Hatfield Consultants	1-week meeting 1-month writing	Hughes and Whittier (2008)
Oregon Water Temperature Standard	Oregon Watershed Enhancement Board	2-day meeting 4-month writing	IMST (2004)
Pebble Dam	Wild Salmon Center Trout Unlimited	1-day meeting 1-year writing	Chambers et al. (2012)
Rio Grande Silvery Minnow	Bureau of Reclamation	1-week meeting 6-month writing	Hubert et al. (2016)
SAB Review of Connectivity Report	USEPA	1-week meeting 6-month writing	SAB (2014)
Salmonid Conservation	National Marine Fisheries Service	3-day meeting 2-year writing	Spence et al. (1996)

steps and more time to the review process, such a review is a crucial advance to make the process more in line with the NEPA. More importantly, rushing a decision is far more damaging than taking additional time to improve the probability of getting it right. In other words, peer review follows the United Nation's emphasis on the precautionary principle in making substantive decisions about the environment (EEA, 2001; Kriebel et al., 2001; DellaSala et al., 2022).

6. Conclusions

The expansive consumption of natural resources and even the transition to renewable energy economies will necessitate new mines, water storage and distribution developments, intensified forest management, and energy projects globally, all of which will have substantial and cumulative impacts. To move towards this future in a just and sustainable way, countries need to carefully assess the social, economic, physical, chemical, and biological risks (e.g., USEPA, 2014) using ISPR and assess accountability in response to ISPR. We conclude that the current EIS process nationally and globally is fundamentally flawed because it lacks accountability. Requiring ISPR would improve EIS credibility as well as both ecological health and economic outcomes. The CEQ and comparable agencies in other nations (Eccleston, 2008) must make these necessary and fundamental changes to the EIS process before embarking on another set of risky projects and management programs.

Correcting the flawed EIS process is a political problem—not a scientific one. Therefore, we urge that the CEQ, similar organizations in other nations, international agencies such as the World Bank and International Seabed Authority, professional societies, and the National Academy of Sciences convene formal review panels regarding how best to increase the role of ISPR in the EIS process. Those reviews should include a thorough, public review of prior EISs, including what went wrong with them and what succeeded, both ecologically and socioeconomically. The EIS processes must be overhauled to incorporate ISPR via conflict-of-interest waivers signed by scientists to assure no connection to project or agency funding sources. It does not serve decision makers or the public well if the results of EISs are misleading, wasteful of time and money, cannot be trusted, and are based on picking sides in scientific disputes in favor of desired outcomes (Lessing and Smosna, 1975; Peterson, 1993; Fairweather, 1994; Ortolano and Shepherd, 1995; Thompson et al., 1997). This is especially true given the current global climate and biodiversity crises (Gannon, 2021; Ripple et al., 2023). Peer review is the basis of all types of good science; thus, peer review should not be circumvented if we are to ensure effective environmental management and the protection of public safety.

CRediT authorship contribution statement

Robert M. Hughes: Writing – review & editing, Writing – original draft, Conceptualization. **David M. Chambers:** Writing – review & editing, Writing – original draft, Conceptualization. **Dominick A. DellaSala:** Writing – review & editing. **James R. Karr:** Writing – review & editing, Writing – original draft. **Susan C. Lubetkin:** Writing – review & editing, Writing – original draft. **Sarah O'Neal:** Writing – review & editing. **Robert L. Vadas:** Writing – review & editing. **Carol Ann Woody:** Writing – review & editing.

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As a Co-editor-in-Chief of *Water Biology & Security*, RMH examines the English and content of some manuscripts, but he was not involved in the manuscript editorial or peer reviews or the decision to publish this article. All authors declare no known competing financial interests or personal relationships that could potentially influence the work reported in this paper.

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References

ACOE, 2019. ACOE (U.S. Army Corps of Engineers). Pebble Project EIS. Draft Environmental Impact Statement. Executive Summary. U.S. Army Corps of Engineers. https://www.arlis.org/docs/vol1/Pebble/Draft-EIS/Executive_summary.pdf.

ACOE, 2020. ACOE (U.S. Army Corps of Engineers). Pebble Project Final Environmental Impact Statement. U.S. Army Corps of Engineers. https://www.arlis.org/docs/vol1/Pebble/Final-EIS.

ADEC (Alaska Department of Environmental Conservation), 2021. Statewide Oil and Hazardous Substance Spills Database, 1995-present. https://dec.alaska.gov/applications/spar/publicmvc/perp/spillsearch.

Adelman, D.E., 2004. Scientific activism and restraint: the interplay of statistics, judgment, and procedure in environmental law. Notre Dame Law Rev. 79, 497–584.

AREMP (Aquatic and Riparian Effectiveness Monitoring Plan), 2023. Interagency Regional Monitoring: Watersheds. https://www.fs.usda.gov/r6/reo/monitoring/watersheds.php.

Ayles, G.B., Dube, M., Roseberg, D., 2004. Oil sands regional aquatic monitoring program (RAMP). Scientific Peer Review of the Five Year Report (1997-2001). Prepared for Alberta Environment and Parks, Environmental Monitoring and Science Division, RAMP (Regional Aquatics Monitoring Program) Steering Committee, Lac la Biche, Canada. http://www.andrewnikiforuk.com/Dirty_Oil_PDFs/RAMP%20Peer%20review.pdf.

Baker, W.L., Hanson, C.T., Williams, M.A., DellaSala, D.A., 2023. Countering omitted evidence of variable historical forests and fire regime in western USA dry forests: the low-severity fire model rejected. Fire 6, 146. https://doi.org/10.3390/fire6040146.

Barron, M.G., Vivian, D.N., Heintz, R.A., Yim, U.H., 2020. Long-term ecological impacts from oil spills: comparison of Exxon Valdez, Hebei Spirit, and Deepwater Horizon. Environ. Sci. Technol. 54, 6456–6467.

Bella, D.A., 1987. Organizations and systemic distortion of information. J. Prof. Issues Bioengin 113, 360–370.

Benkendorff, K., 1999. The need for more stringent requirements in Environmental Impact Assessment: shell Cove Marina case study. Pac. Conserv. Biol. 5, 214–223.

BOEM, 2018. 2019-2024 National Outer Continental Shelf Oil and Gas Leasing Draft Proposed Program.

Bowker, L.N., 2021. Potential risk index for any tailings portfolio or facility: a tool for identifying and classifying potentially at-risk TSFs. World Mine Tailings Failures.

Bowker, L.N., Chambers, D.M., 2017. The dark shadow of the supercycle: tailings failure risk & public liability reach all-time highs. Environments 4 (4). https://doi.org/10.3390/environments404007.

Brasil, 2012. Lei No 12.651 de 12 de maio de 2012. http://www.planalto.gov.br/ccivil_03/_ato2011-2014/2012/lei/l12651.htm.

Buckley, R., 1991. What's wrong with EIA? Search 20, pp. 146–147.

Buckley, R., 1991. How accurate are environmental impact predictions? Ambio 20, 161–162.

Byrne, P., Hudson-Edwards, K.A., Bird, G., Macklin, M.G., Brewer, P.A., Williams, R.D., Jamieson, H.E., 2018. Water quality impacts and river system recovery following the 2014 Mount Polley mine tailings dam spill, British Columbia, Canada. Appl. Geochem. 91, 64–74.

Callisto, M., et al., 2014. In: Hughes, R.M., Lopes, J.M., Castro, M.A. (Eds.), Ecological Conditions in Hydropower Basins. Peixe Vivo Series 2. Companhia Energética de Minas Gerais. Belo Horizonte. Minas Gerais, Brazil.

Canadian impact assessment act. https://laws.justice.gc.ca/eng/acts/i-2.75/index.html, 2019.

CEAA (Canadian Environmental Assessment Agency), 2023. Overview of the impact assessment act. https://www.canada.ca/content/dam/iaac-acei/documents/mandate/president-transition-book-2019/overview-impact-assessment-act.pdf.

CEC (Copperhead Environmental Consulting), 2022. Rainy River Watershed Withdrawal: Case Studies Report. U.S. Department of Agriculture. https://eplanning.blm.gov/public_projects/2022642/200540165/20071351/250077533/Rainy%20River%20Withdrawal%20-%20Case%20Studies%20Report.pdf.

CEQ (Council on Environmental Quality), 2021. A citizen's guide to NEPA: having your voice heard. https://www.energy.gov/nepa/articles/citizens-guide-nepa-having-your-voice-heard-ceq-2007-revised-2021.

Chambers, D., Moran, R., Trasky, L., Bryce, S., Danielson, L., Fulkerson, L., Goin, J., Hughes, R.M., Konigsberg, J., Spies, R., Thomas, G., Trenholm, M., Wigington, T., 2012. Bristol Bay's Wild Salmon Ecosystems and the Pebble Mine: Key Considerations for a Large-Scale Mine Proposal. Wild Salmon Center and Trout Unlimited: Portland, Oregon, USA.

Chang, T., Nielsen, E., Auberle, W., Solop, F.I., 2013. A quantitative method to analyze the quality of EIA information in wind energy development and bat/avian assessments. Environ. Impact Assess. Rev. 38, 142–150.

Collard, R.C., Dempsey, J., 2022. Future eco-perfect: temporal fixes of liberal environmentalism. Antipode 54, 1545–1565.

Collard, R.-C., Dempsey, J., Muir, B.R., Allan, R., Herd, A., Bode, P., 2023. Years late and millions short: a predictive audit of economic impacts for coal mines in British Columbia, Canada. Environ. Impact Assess. Rev. 100. https://doi.org/10.1016/j.eiar.2023.107074.

Daniel, W.M., Infante, D.M., Hughes, R.M., Esselman, P.C., Tsang, Y.-P., Wieferich, D., Herreman, K., Cooper, A.R., Wang, L., Taylor, W.W., 2014. Characterizing coal and mineral mines as a regional source of stress to stream fish assemblages. Ecol. Indicat. 50, 50–61.

Davidson, S., 2023. First of the Klamath dams comes down. Trout Unlimited. https://www.tu.org/magazine/conservation/barriers/dam-removal/first-of-the-klamath-dams-comes-down.

DellaSala, D.A., Baker, R., Heiken, D., Frissell, C.A., Karr, J.R., Nelson, S.K., Noon, B.R., Olson, D., Strittholt, J., 2015. Building on two decades of ecosystem management and biodiversity conservation under the Northwest Forest Plan, USA. Forests 6, 3326–3352.

DellaSala, D.A., Baker, B.C., Hanson, C.T., Ruediger, L., Baker, W., 2022. Have western USA fire suppression and megafire active management approaches become a contemporary Sisyphus? Biol. Conserv. 268. https://doi.org/10.1016/j.biocon.2022.109499.

Dillon, L., Sellers, C., Underhill, V., Shapiro, N., Ohayon, J.L., Sullivan, M., Brown, P., Harrison, J., Wylie, S., 2018. The Environmental Protection Agency in the Early Trump Administration: Prelude to Regulatory Capture. Amer. J. Publ. Health. https://ajph.aphapublications.org/doi/pdf/10.2105/AJPH.2018.304360.

Duncan, R., 2008. Problematic practice in integrated impact assessment: the role of consultants and predictive computer models in burying uncertainty. Impact Assess. Proj. Apprais. 26, 53–66.

Dunham, J., 19 coauthors, 2023. Northwest Forest Plan—the first 25 years (1994-2018): watershed condition status and trends. Gen. Tech. Rep. PNW-GTR-1010. U.S. Forest Service, Portland, USA. https://doi.org/10.2737/PNW-GTR-1010.

Eccleston, C.H., 2008. NEPA and Environmental Planning: Tools, Techniques, and Approaches for Practitioners. CRC Press, Boca Raton, USA.

ECONorthwest, 2019. Lower Snake River Dams: Economic Tradeoffs of Removal. Seattle, USA. https://static1.squarespace.com/static/597fb96acd39c34098e8d423/t/5d41bbf522405f0001c67068/1564589261882/LSRD_Economic_Tradeoffs_Report.pdf.

EEA (European Environment Agency), 2001. Late lessons from early warnings: the precautionary principle 1896-2000. Environmental issue report 22. https://www.eea.europa.eu/publications/environmental_issue_report_2001_22.

EFI+ CONSORTIUM, 2009. Manual for the application of the new European Fish Index –EFI+. A fish-based method to assess the ecological status of European running waters in support of the Water Framework Directive. https://www.researchgate.net/publication/40434892_Manual_for_the_application_of_the_new_European_Fish_Index_-_EFI_A_fish-based_method_to_assess_the_ecological_status_of_European_running_waters_in_support_of_the_Water_Framework_Directive.

Elliott, E.D., 2003. Strengthening science's voice at EPA. Law contemp. Probl 46, 45–62. https://openyls.law.yale.edu/handle/20.500.13051/4633.

Escobar, H., 2015. Mud tsunami wreaks ecological havoc in Brazil. Science 350, 1138–1139.

EU (European Union), 2014. Directive 2014/52/EU of the European Parliament and of the Council of 16 April 2014 amending Directive 2011/92/EU on the assessment of the effects of certain public and private projects on the environment Text with EEA relevance. https://eur-lex.europa.eu/legal-content/EN/TXT/?uri=celex%3A32014L0052.

Fairweather, P.G., 1989. Where is the science in EIA? Search 20, pp. 141–144.

Fairweather, P.G., 1994. Improving the use of science in environmental assessments. Aust. Zool. 29, 217–223.

FEMAT (Forest Ecosystem Management Assessment Team), 1993. Forest Ecosystem Management: an Ecological, Economic, and Social Assessment. U.S. Forest Service, U.S. Fish & Wildlife Service, National Marine Fisheries Service. National Park Service, U.S. Bureau of Land Management, U.S. Environmental Protection Agency, Washington, USA.

FERC (Federal Energy Regulatory Commission), 2022. Final environmental impact statement for hydropower license surrender and decommissioning for the lower Klamath project. https://elibrary.ferc.gov/eLibrary/filelist?accession_number=20220826-3006&optimized=false.

Friel, C., Leary, T., Norris, H., Waford, R., Sargent, B., 1993. GIS tackles oil spill in Tampa Bay. GIS World 6 (11), 30–33.

Frissell, C.A., Baker, R.J., DellaSala, D.A., Hughes, R.M., Karr, J.R., McCullough, D.A., Nawa, R.K., Scurlock, M.C., Wissmar, R.C., 2014. Conservation of Aquatic and Fishery Fesources in the Pacific Northwest: Implications of New Science for the Aquatic Conservation Strategy of the Northwest Forest Plan. Coast Range Association, Corvallis (Oregon).

Gaines, W.L., Hessburg, P.F., Aplet, G.H., Henson, P., Prichard, S.J., Churchill, D.J., Jones, G.M., Isaak, D.J., Vynne, C., 2022. Climate change and forest management on federal lands in the Pacific Northwest, USA: managing for dynamic landscapes. For. Ecol. Manage. https://doi.org/10.1016/j.foreco.2021.119794.

Gannon, P., 2021. The time is now to improve the treatment of biodiversity in Canadian environmental impact statements. Environ. Impact Assess. Rev. 86. https://doi.org/10.1016/j.eiar.2020.106504.

Gordon, N.D., McMahon, T.A., Finlayson, B.L., 1992. Stream Hydrology: an Introduction for Ecologists. John Wiley & Sons, New York, USA.

Gresh, T., Lichatowich, J., Schoonmaker, P., 2000. An estimation of historic and current levels of salmon production in the northeast Pacific ecosystem: evidence of a nutrient deficit in the freshwater systems of the Pacific Northwest. Fisheries 25, 15–21.

Harwood, D.W., Russell, E.R., 1990. Present Practices of Highway Transportation of Hazardous Materials. US Department of Transportation, Federal Highway Administration. May 1990.

Henderson, R.C., Archer, E.K., Bouwes, B.A., Coles-Ritchie, M.S., Kershner, J.L., 2005. PACFISH/INFISH biological opinion (PIBO): effectiveness monitoring program seven-year status report 1998 through 2004. Gen. Tech. Rep. RMRS-GTR-162. U.S. Forest Service. Fort Collins, USA. https://www.google.com/books/edition/PACFISH_INFISH_Biological_Opinion_PIBO/yI5-I5m0qKsC?hl=en&gbpv=1&pg=PP1&printsec=frontcover.

Henjum, M.G., Karr, J.R., Bottom, D.L., Perry, D.A., Bednarz, J.C., Wright, S.G., Beckwitt, S.A., Beckwitt, E., 1994. Interim Protection for Late-Successional Forests, Fisheries, and Watersheds: National Forests East of the Cascade Crest, Oregon and Washington. The Wildlife Society, Bethesda, USA.

Hilborn, R., Walters, C., 1981. Pitfalls of environmental baseline and process studies. Environ. Impact Assess. Rev. 2, 265–278.

Hill, R.A., Moore, C.C., Doyle, J.M., Leibowitz, S.G., Ringold, P.L., Rashleigh, B., 2023. Estimating biotic integrity to capture existence value of freshwater ecosystems. Proc. Natl. Acad. Sci. U.S.A. https://doi.org/10.1073/pnas.2120259119.
Hubert, W., Fabrizio, M., Hughes, R.M., Cusack, M., 2016. Summary of Findings by the External Expert Panelists: Rio Grande Silvery Minnow Population Monitoring Workshop. U.S. Bureau of Reclamation, Albuquerque, USA.
Hughes, R.M., 1993. Stream Indicator and Design Workshop. U.S. Environmental Protection Agency, Corvallis, OR. EPA/600/R-93/138.
Hughes, R.M., Vadas, R.L., 2021. Agricultural effects on streams and rivers: a western USA focus. Water 13. https://doi.org/10.3390/w13141901.
Hughes, B., Whittier, T.R., 2008. An Assessment of the Regional Aquatics Monitoring Program (RAMP) Fish Survey. Hatfield Consultants (Vancouver, Canada).
Hughes, R.M., Paulsen, S.G., Stoddard, J.L., 2000. EMAP-Surface Waters: a national, multiassemblage, probability survey of ecological integrity. Hydrobiol. 422/423, 429–443.
Hughes, R.M., Amezcua, F., Chambers, D.M., Daniel, W.M., Franks, J.S., Franzin, W., MacDonald, D., Merriam, E., Neall, G., Pompeu, P.S., Reynolds, L., Woody, C.A., 2016. AFS position paper and policy on mining and fossil fuel extraction. Fisheries 41, 12–15.
Hughes, R.M., Vadas, R.L., Michael Jr., J.H., Knutson Jr., A.C., DellaSala, D.A., Burroughs, J., Beecher, H., 2021. Why advocate—and how? In: DellaSala, D.A. (Ed.), Conservation Science and Advocacy for a Planet in Crisis: Speaking Truth to Power. Elsevier, New York, USA, pp. 177–197.
Hughes, R.M., Karr, J.R., Vadas, R.L., DellaSala, D.A., Callisto, M., Feio, M.J., Ferreira, T., Kleynhans, N., Ruaro, R., Yoder, C.O., Michael, J.H., 2023. Global concerns related to water biology and security: the need for language and policies that safeguard living resources versus those that dilute scientific knowledge. Wat. Biol. Secur., 100191 https://doi.org/10.1016/j.watbs.2023.100191.
IMST (Independent Multidisciplinary Science Team), 2004. Oregon's Water Temperature Standard and its Application: Causes, Consequences, and Controversies Associated with Stream Temperature. https://ir.library.oregonstate.edu/concern/technical_reports/zp38wc69f.
Incardona, J.P., Carls, M.G., Holland, L., Linbo, T.L., Baldwin, D.H., Myers, M.S., Peck, K.A., Tagal, M., Rice, S.D., Scholz, N.L., 2015. Very low embryonic crude oil exposures cause lasting cardiac defects in salmon and herring. Sci. Rep. 5, 13499.
Ji, Z.-G., Johnson, W.R., Wikel, G., 2014a. Catastrophic oil spill analysis. Risk Analysis XI, WIT Trans. Inform. Commun. Technol. 47, 17–25.
Ji, Z.-G., Johnson, W.R., Wikel, G.L., 2014b. Statistics of extremes in oil spill risk analysis. Environ. Sci. Technol. 48, 10505–10510.
Ji, Z.-G., Li, Z., Johnson, W.R., Auad, G., 2021. Progress of the oil spill risk analysis (OSRA) model and its applications. J. Mar. Sci. Eng. 9 (2), 195. https://doi.org/10.3390/jmse9020195.
Joly, J.L., Reynolds, J.H., Robards, M., 2010. Recognizing when the best scientific data available isn't. Stanford Environ. Law J. 29, 247–282.
Karr, J.R., 1991. Biological integrity: a long-neglected aspect of water resource management. Ecol. Appl. 1, 66–84.
Karr, J.R., Chu, E.W., 1999. Restoring Life in Running Waters: Better Biological Monitoring. Island Press, Washington DC, USA.
Karr, J.R., Dudley, D.R., 1981. Ecological perspective on water quality goals. Environ. Manag. 5, 55–68.
Karr, J.R., Larsen, E.R., Chu, E.W., 2022. Ecological integrity is both real and valuable. Conserv. Sci. Pract. https://doi.org/10.1111/csp2.583.
Knutson, K.L., Naef, V.L., 1997. Management recommendations for Washington's priority habitats: riparian. Washington Department of Fish and Wildlife, Olympia, USA. https://wdfw.wa.gov/sites/default/files/publications/00029/wdfw00029.pdf.
Kriebel, D., Tickner, J., Epstein, P., Lemons, J., Levins, R., Loechler, E.L., Quinn, M., Rudel, R., Schettler, T., Stoto, M., 2001. The precautionary principle in environmental science. Environ. Health Perspect. 109, 871–876.
Kuipers, J.R., Maest, A.S., MacHardy, K.A., Lawson, G., 2006. Comparison of predicted and actual water quality at hardrock mines, the reliability of predictions in environmental impact statements. Earthworks, Washington, USA. https://earthworks.org/files/publications/ComparisonsReportFinal.pdf.
Lees, J., Jaeger, J.A.G., Gunn, J.A.E., Noble, B.F., 2016. Analysis of uncertainty consideration in environmental assessment: an empirical study of Canadian EA practice. J. Environ. Plan. Manag. 59, 2024–2044.
Lessing, P., Smosna, R.A., 1975. Environmental impact statements – worthwhile or worthless? Geol. 3, 241–242.
Loftin, M.K., Toth, L.A., Obeysekera, J.T.B. (Eds.), 1988. Kissimmee River Restoration Symposium. South Florida Water Management District (Orlando, USA).
Lubetkin, S.C., 2020. The tip of the iceberg: three case studies of spill risk assessments used in environmental impact statements. Mar. Pollut. Bull. 152, 110613. https://doi.org/10.1016/j.marpolbul.2019.110613.
Lubetkin, S.C., 2022. Alaska mining spills: a comparison of the predicted impacts described in permitting documents and spill records from five major operational hardrock mines. https://earthworks.org/resources/alaska-mining-spills/.
Maas-Hebner, K.G., Schreck, C.B., Hughes, R.M., Yeakley, J.A., Molina, N., 2016. Scientifically defensible fish conservation and recovery plans: addressing diffuse threats and developing rigorous adaptive management plans. Fisheries 41, 276–285.
McGarvey, D.J., 2007. Merging precaution with sound science under the endangered species act. BioScience 57, 65–70. https://doi.org/10.1641/B570110.
McGarvey, D.J., Marshall, B., 2005. Making sense of scientists and "sound science": truth and consequences for endangered species in the Klamath Basin and beyond. Ecol. Law Q. 32, 73–111.
Meretsky, V.J., Fischman, R.L., Karr, J.R., Ashe, D.M., Scott, M.J., Noss, R.F., Schroeder, R.L., 2006. New directions in conservation for the national wildlife refuge system. BioScience 56, 135–142.
Mildrexler, D.J., Berner, L.T., Law, B.E., Birdsey, R.A., Moomaw, W.R., 2023. Protect large trees for climate mitigation, biodiversity, and forest resilience. Cons. Sci. Prac. https://doi.org/10.1111/csp2.12944.
NEPA (National Environmental Policy Act): https://www.epa.gov/nepa/what-national-environmental-policy-act.
Noble, B.F., 2020. Introduction to Environmental Impact Assessment: a Guide to Principles and Practice. Oxford University Press, Toronto, Canada.
Olson, D.H., Anderson, P.D., Frissell, C.A., Welsh Jr., H.H., Bradford, D.F., 2007. Biodiversity management approaches for stream-riparian areas: perspectives for Pacific Northwest headwater forests, microclimates, and amphibians. For. Ecol. Manage. 246, 81–107.
Ortolano, L., Shepherd, A., 1995. Environmental impact assessment: challenges and opportunities. Impact Assess 13, 3–30. https://doi.org/10.1080/07349165.1995.9726076.
Perneger, T.V., Combescure, C., 2017. The distribution of *P*-values in medical research articles suggested selective reporting associated with statistical significance. J. Clin. Epidemiol. 87, 70–77.
Peterson, C.H., 1993. Improvement of environmental impact analysis by application of principles derived from manipulative ecology: lessons from coastal marine case histories. Aust. J. Ecol. 18, 21–52.
Peterson, C.H., Rice, S.D., Short, J.W., Esler, D., Bodkin, J.L., Ballachey, B.E., Irons, D.B., 2003. Long-term ecosystem responses to the Exxon Valdez oil spill. Science 19, 282–286.
Piatt, J.F., Ford, R.G., 1996. How many seabirds were killed by the Exxon Valdez oil spill? Am. Fish. Soc. Symp. 18, 712–719.
PIBOMP (PacFish/InFish Biological Opinion Monitoring Program), 2023. PacFish/InFish overview. https://www.fs.usda.gov/detail/r4/landmanagement/resourcemanagement/?cid=stelprd3845865.
Possingham, H.P., Andelman, S.J., Noon, B.R., Trombulak, S., Pulliam, H.R., 2001. Making smart conservation decisions. In: M.E. Soule, M.E., Orians, G.H. (Eds.), Conservation Biology: Research Priorities for the New Decade. Island Press, Washington, USA, pp. 225–244.
Quinn, T., Wilhere, G.F., Krueger, K.L. (Eds.), 2020. Riparian Ecosystems, Volume 1: Science Synthesis and Management Implications. Washington Department of Fish and Wildlife, Olympia, USA.
Rau, B., 2017. Process design: voluntary clean water guidance for agricultural activities. Washington Department of Ecology. Water Qual. Prof. Olympia, USA. https://ecology.wa.gov/getattachment/096726e8-5df2-4d33-b958-d1249063d9ad/ProcessDesign-Final.pdf.
Ripple, W.J., Wolf, C., Gregg, J.W., Rockstrom, J., Newsome, T.M., Law, B.E., Marques, L., Lenton, T.M., Xu, C., Huq, S., Simons, L., King, S.D.A., 2023. The 2023 state of the climate report: entering uncharted territory. BioScience. https://doi.org/10.1093/biosci/biad080.
Rockweit, J.T., 35 coauthors, 2023. Range-wide sources of variation in reproductive rates of northern spotted owls. Ecol. Indicat. 33. https://doi.org/10.1002/eap.2726.
SAB, 2014. SAB review of the draft EPA report *Connectivity of streams and wetlands to wownstream waters: a review and synthesis of the scientific evidence*. U.S. Environmental Protection Agency, Washington, USA. EPA-SAB-15-001.
Salvador, G.N., Leal, C.G., Brejão, G.L., Pessali, T.C., Alves, C.B.M., Rosa, G.R., Ligeiro, R., de Assis Montag, L.F., 2020. Mining activity in Brazil and negligence in action. Perspect. Ecol. Conserv. 18, 139–144. https://doi.org/10.1016/j.pecon.2020.05.003.
Schindler, D.W., 1976. The impact statement boondoggle. Science 192, 509.
Sheaves, M., Coles, R., Dale, P., Grech, A., Pressey, R.L., Waltham, N.J., 2016. Enhancing the value and validity of EIA: serious science to protect Australia's Great Barrier Reef. Conserv. Lett. 9, 377–383.
Spence, B.C., Lomnicky, G.A., Hughes, R.M., Novitzki, R.P., 1996. An Ecosystem Approach to Salmonid Conservation. TR-4501-96-6057. National Marine Fisheries Service, Portland, USA.
Stern, N., 2013. The structure of economic modeling of potential impacts of climate change: grafting gross underestimation of risk onto already narrow science models. J. Econ. Lit. 51, 838–859.
Sweeney, B.W., Newbold, J.D., 2014. Streamside forest buffer width needed to protect stream water quality, habitat, and organisms: a literature review. J. Am. Water Resour. Assoc. 50, 560–584.
Thompson, S., Treweek, J.R., Thurling, D.J., 1997. The ecological component of environmental impact assessment: a critical review of British environmental statements. J. Environ. Plann. Manag. 40, 157–171.
Treweek, J., 1996. Ecology and environmental impact assessment. J. Appl. Ecol. 33, 191–199.
USEPA (U.S. Environmental Protection Agency), 2000. Liquid Assets: America's Water Resources at a Turning Point. EPA-840, Washington, USA.
USEPA (U.S. Environmental Protection Agency), 2004. Nationwide identification of hardrock mining sites. Evaluation Report. Report 2004-P-00005. Office of Inspector General. USEPA, Washington, USA.
USEPA (U.S. Environmental Protection Agency), 2014. An Assessment of Potential Mining Impacts on Salmon Ecosystems of Bristol Bay, Alaska. U.S. Environmental Protection Agency, Washington, USA, 910-R-14-001A. https://www.epa.gov/bristolbay/bristol-bay-assessment-final-report-2014.
USEPA (U.S. Environmental Protection Agency), 2019. Letter to Shane McCoy, USACOE-Alaska District. https://www.epa.gov/sites/default/files/2019-07/documents/epa-comments-draft-eis-pebble-project-07-01-2019.pdf.
USEPA (U.S. Environmental Protection Agency), 2023a. What is the national environmental policy act? https://www.epa.gov/nepa/what-national-environmental-policy-act.
USEPA (U.S. Environmental Protection Agency), 2023b. Final Determination of the U.S Environmental Protection Agency Pursuant to Section 404(C) of the Clean Water Act,

Pebble Deposit Area, Southwest Alaska. Office of Water, Washington DC. https://www.epa.gov/system/files/documents/2023-01/Pebble-Deposit-Area-404c-FD-Jan2023.pdf.

USG (US Government), 2023. National environmental policy act of 1969. Public law 91–190 (as amended through P.L. 118–5). https://www.govinfo.gov/content/pkg/COMPS-10352/uslm/COMPS-10352.xml.

Utts, J., 2021. Enhancing data science ethics through statistical education and practice. Int. Stat. Rev. 89, 1–17. https://doi.org/10.1111/insr.12446.

Vadas Jr., R.L., Hughes, R.M., Bello-Gonzales, O., Callisto, M., Carvalho, D., Chen, K., Davies, P.E., Ferreira, M.T., Fierro, P., Harding, J.S., Kleynhans, C.J., Macedo, D.R., Mercado-Silva, N., Moya, N., Nichols, S.J., Pompeu, P.S., Ruaro, R., Stevenson, R.J., Terra, B.F., Thirion, C., Ticiani, D., Yoder, C.O., 2022. Assemblage-based biomonitoring of freshwater ecosystem health via multimetric indices: a critical review and suggestions for improving their applicability. Wat. Biol. Secur. 1 (3), 100054. https://doi.org/10.1016/j.watbs.2022.100054.

Virgilio, C.S., Lacerda, D., de Oliveira, B.C.V., Sartori, E., Campos, G.M., Pereira, A.L.S., de Aguiar, D.B., Souza, T.S., de Almeida, M.G., Thompson, F., de Rezende, C.E., 2020. Metal concentrations and biological effects from one of the largest mining disasters in the world (Brumadinho, Minas Gerais, Brazil). Sci. Rep. 10. https://doi.org/10.1038/s41598-020-62700-w.

Wasserstein, R.L., Lazar, N.A., 2016. The ASA statement on *p*-values: context, process, and purpose. Am. Statistician 70, 129–133.

Whittaker, K., Goldman, P., 2021. Shifting the burden of proof to minimize impacts during the science-policy process. In: DellaSala, D.A. (Ed.), Conservation Science and Advocacy for a Planet in Crisis: Speaking Truth to Power. Elsevier, New York, USA, pp. 249–274.

Woody, C.A., 2018. Bristol Bay Alaska: Natural Resources of the Aquatic and Terrestrial Ecosystems. J. Ross Publishing, Fort Lauderdale, USA.

Woody, C.A., Hughes, R.M., Wagner, E.J., Quinn, T.P., Roulsen, L.H., Martin, L.M., Griswold, K., 2010. The U.S. General Mining Law of 1872: change is overdue. Fisheries 35, 321–331.

Youngblood, A., 2001. Old-growth forest structure in Eastern Oregon and Washington. Northwest Sci. 75, 110–118.

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Distinct fluvial and adfluvial migration patterns of a relict charr, *Salvelinus confluentus*, stock in a mountainous watershed, Idaho, USA

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Abstract – Sixty-five large (>385 mm fork length) bull trout (*Salvelinus confluentus*), a threatened relict charr (Family Salmonidae), were captured in the upper East Fork South Fork Salmon River (EFSFSR), Idaho, USA and implanted with radio tags to investigate their spatial and temporal movements and distribution throughout the South Fork Salmon River (SFSR) basin and beyond. All radio-tagged fish were migratory. Most fish displayed a fluvial migration pattern. They typically overwintered in the larger rivers downriver of the EFSFSR (SFSR and the Salmon River further downstream), migrated upriver to the EFSFSR in June and further upriver into small (<2 m wide) tributaries to spawn in August and September. Both consecutive-year and nonconsecutive-year spawners were found. A typical migration distance between the overwintering habitat and the spawning habitat was 100 km. A minor fraction (<10%) of the fish displayed an adfluvial life history pattern, overwintering in a small (2 ha) 60-year-old flooded mine pit in the EFSFSR upstream of the spawning tributaries. The stock exhibited distinct site fidelity for spawning and overwintering. Similar fluvial and adfluvial migration patterns have been reported for bull trout in the region as well as for other charr species worldwide. Effective management of this and other migratory charr stocks will require protection of a wide range of habitats, from large rivers to the smallest tributaries.

Key words: Charr; migration; adfluvial; fluvial; bull trout

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Introduction

Effective conservation of charrs (*Salvelinus* spp.) should be based on a thorough understanding of their often complex life histories and migration patterns (Rieman & McIntyre 1993). Throughout the northern hemisphere, charrs display a wide range of migration patterns, including migrations between river segments (fluvial potamodromy; Schill et al. 1994), between rivers and lakes or reservoirs (adfluvial; Stelfox 1997) and between rivers or lakes and the sea (anadromy; Johnson 1989; Berg & Berg 1993). Migrations, both upstream and downstream, are usually undertaken for spawning (Schill et al. 1994) or for improved feeding opportunities (e.g., Naslund 1992; Doucett et al. 1999; Gulseth & Nilssen 2001). The great plasticity in charr life histories (Reist 1989), including the development of diverse migration patterns (McCart 1980) and exploratory behaviours, has been suggested as being highly adaptive for energy acquisition in the habitats

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in which the species evolved: cold, unproductive, ice- and sediment-influenced, unstable waters associated with glaciation (Power 2002).

The bull trout (*Salvelinus confluentus*), a native charr of the northwestern United States and Canada, has suffered major declines in the past century, especially those relict stocks inhabiting the southern portion of its range. Several factors have contributed to the decline of the species, including overharvest, habitat disruption, non-native species competition and hybridisation with non-native brook trout (*Salvelinus fontinalis*). Management of bull trout has been complicated by several factors, including the species' patchy distribution (Rieman & McIntyre 1993), its tendency to migrate long distances and utilise widely separated habitats (Swanberg 1997) and our lack of knowledge about population (stock) identity and discreteness (Rieman & McIntyre 1993). Insufficient knowledge of the spatial and temporal distribution of individual charr stocks also makes it difficult to designate appropriate conservation areas (Stowell et al. 1996). All the above problems exist for the bull trout in the East Fork South Fork Salmon River (EFSFSR), Idaho, a southern relict stock thought to be part of a central Idaho stronghold for this threatened species (USFWS 1998). As with many charr stocks in Idaho, the Pacific Northwest, and worldwide, the distribution and migratory range of the EFSFSR bull trout stock is poorly understood. The objectives of this study were to: (i) assess spatial and temporal distribution of bull trout in the upper EFSFSR and its tributaries; and (ii) characterise individual and group bull trout movements by season. To the extent that migration patterns of bull trout in EFSFSR correlate with those of other charrs, results from and implications of this study will be useful in the management of various migratory charr stocks worldwide.

Study area

The study was located in west-central Idaho, USA within the South Fork Salmon River (SFSR) drainage. The study area has a short growing season, with most precipitation (mean 79 cm·year^{-1}) falling as snow. Frost can occur any day of the year at elevations higher than 2133 m (Hogen 2002).

Fish sampling was centred in the upper EFSFSR watershed (area 33,994 ha). The EFSFSR flows through a forested, v-shaped canyon with mostly steep topography. EFSFSR stream channel gradients average about 4%. Geology in the SFSR is primarily granitic (Idaho batholith) with some volcanic and metamorphic material. The granitic material results in a low productivity of the streams; it also weathers into a fine substrate most common in disturbed streams (Klamt 1976; Clayton & Megahan 1997). The highest runoff period in the EFSFSR is typically in a 6-week period in May and June from snowmelt. Over the period 1929–1995, mean June discharge of the EFSFSR at Stibnite was 3.3 m^3·s^{-1}; mean August discharge was 0.45 m^3·s^{-1} (Kuzis 1997).

Twentieth century gold-mining activities have resulted in the formation of the Glory Hole, a former mine pit and now a 2-ha body of water located 19 km upstream of the EFSFSR mouth. The Glory Hole had a maximum depth of 13.4 m in 1999, with most of the pit deeper than 6 m.

Methods

Capture

Bull trout were captured in the EFSFSR watershed (above the confluence with the SFSR) by hook and line sampling using artificial lures as well as circle hooks baited with salmon eggs. Sampling occurred almost daily from 29 June to 12 August 1999, and from 3 July to 15 August 2000. The 65 bull trout chosen to be radio tagged ranged in length from 320 to 790 mm fork length (FL) and in weight from 385 to 4390 g.

Radio tagging

Fish were implanted with coded radio tags by surgery. The tags were of three different dimensions designed to keep the tag weight below 2% of the fish's body weight (Winter 1996; Swanberg 1997). In 1999, 25 larger Lotek MCFT-3FM tags (10.3 g, 11 mm × 59 mm) and 11 smaller Lotek MCFT-3BM tags (7.7 g, 11 mm × 43 mm dimensions) were implanted. The typical life for both of these tag types was 238 days, or only a single spawning and overwintering season. In 2000, 11 intermediate-sized tags (Lotek MCFT-3EM; 8.9 g, 11 mm × 49 mm) were implanted. The estimated life of these tags was 439 days. Tags transmitted 24 h·day^{-1} with a 5-s burst rate.

Captured fish were held in individual stream tubes (90 cm × 15 cm or 90 cm × 10 cm) made of water-pervious (2-cm holed) PVC pipe with a sliding door at one end. On warm and sunny days, surgery was postponed until early evening.

Surgical procedures are detailed in Hogen (2002) and summarised here. All surgeries were conducted on site. Immediately prior to surgery, each fish was placed into a holding tank that contained 80–90 mg·l^{-1} tricaine methanesulphonate (MS-222) solution. Anaesthesia occurred in 1–2 min. Each fish was then placed on its dorsum on a padded v-shaped holder. MS-222 solution was continuously pumped over its gills and head to maintain anaesthesia throughout the surgery.

A 4-cm incision was made with a scalpel and scalpel guide anterior to the pelvic fins approximately 3 cm from the mid-ventral line on either the left or right side. A grooved receiver was inserted posteriorly through the incision, with its end positioned posteriorly to the pelvic fins. A 14-gauge needle (10-cm long) was then pushed through the skin onto the grooved receiver and then slid anterior toward the incision. The tag antenna was guided through the needle until it extruded. The needle and grooved receiver were removed from the incision and the tag was placed into the body cavity. Suturing with absorbable sutures consisted of three surgeon's knots.

After tag implantation, each fish was transported back to the stream tube to recover. Fish were held in the stream tube for a minimum of 15 min for recovery and then released at the capture site.

Radio telemetry

Movements of tagged fish were followed by fixed and mobile tracking using two Lotek SRX400 radio receivers, one fixed at a station and the other portable. The fixed receiver was located on a streambank of the SFSR 0.8 km downstream of the Secesh River confluence (Fig. 1). A six-element Yagi directional antenna was mounted on a large tree at the site. A data logger recorded the time and date when radio-tagged fish passed the site. The receiver was removed from November through mid June to avoid freezing temperatures. The portable receiver was moved primarily by truck, but also by airplane and on foot. Mobile tracking was conducted at weekly intervals from July through September, and bimonthly from October through June. During the July through September (3-month) period, each fish was contacted by either fixed or mobile tracking an average of once every 5.6 days in 1999 and once every 4.2 days in 2000. The most frequent average rate of encounter for an individual fish was once every 4.0 days in 1999 and once every 1.9 days in 2000; the least frequent rate of encounter for an individual fish was once every 8.3 days in 1999 and once every 15.2 days in 2000.

Determining the location of a fish within 50–100 m was considered adequate for the study objectives, except during spawning, when more precise locations (to within 1 m) were sought. At suspected spawning sites and times, triangulation was used to pinpoint the fish's location more accurately. Fish locations were recorded using a combination of Global Positioning System (GPS) coordinates, topographic maps or road distances from known locations. When an individual fish was not located by tracking with a truck, tracking by foot was conducted in areas without roads. If a fish was still not located, tracking by aircraft was conducted.

Attempts to observe and locate fish spawning sites were made with each contact during the fall season,

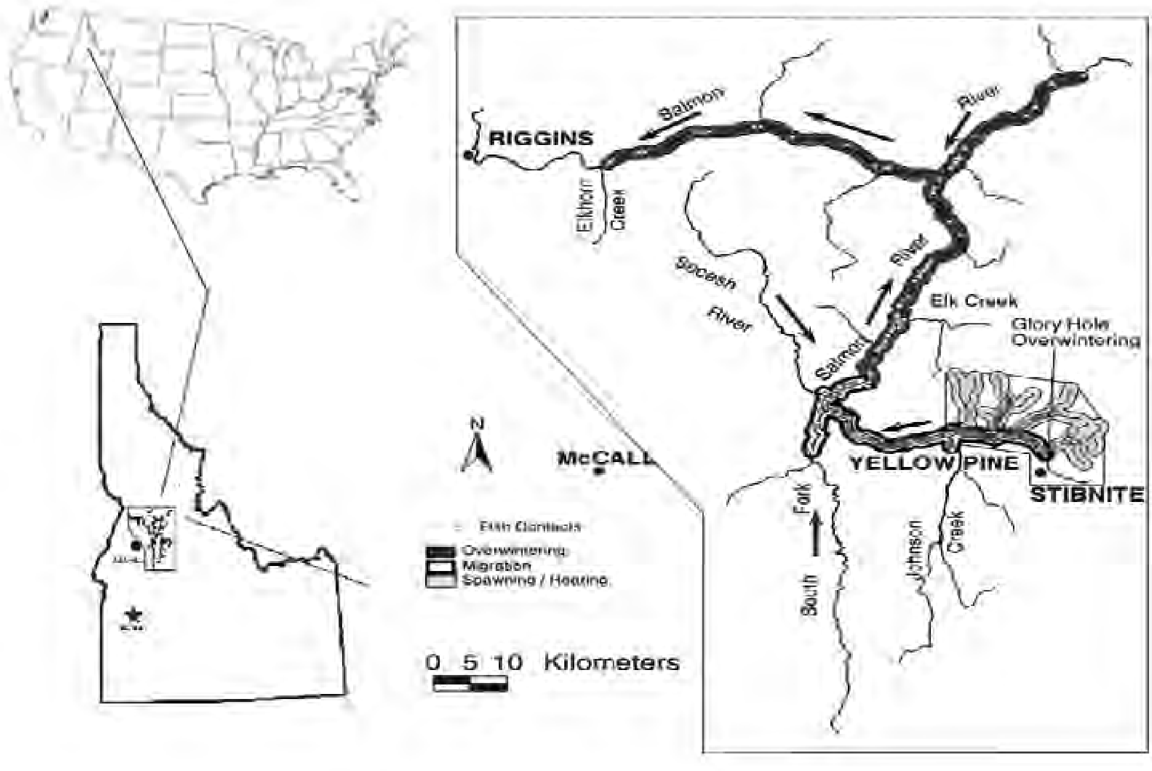

Fig. 1. Map of study area showing bull trout overwintering, migration corridors and spawning and early rearing locations determined by radio telemetry, 1999–2000. Arrows indicate direction of flows.

which was known to be the spawning period from other studies elsewhere (Shepard et al. 1984; Schill et al. 1994; Swanberg 1997). Active redd construction, pairing of fish in small headwater tributaries in the fall and the guarding of a redd were all considered to be evidence of spawning activities.

The telemetry data were quantified by river kilometre, expressed as the distance from the mouth of the Columbia River to the location of the radio-tagged fish. When upstream and downstream movements were >0.2 km between contacts, they were recorded as new locations; if no movement >0.2 km occurred, the fish was considered stationary. In the telemetry analysis, each individual fish was the experimental unit (Winter 1996). Migration was defined as the act of moving from one spatial unit (e.g., overwintering habitat or spawning habitat) to another, and changing position within a single spatial unit was defined as a movement (Baker 1978).

Analysis

All fish locations were graphed and visually inspected for patterns of individual and group movements. Patterns of movement were qualitatively categorised as migratory or nonmigratory. Migratory patterns were further described as fluvial or adfluvial.

To quantitatively test if fish movements were nondirectional or directional, a nonparametric runs test (Ramsey & Schafer 1997) was used for each individual fish and all fish were grouped together from 1999 and 2000. If the fish moved upstream from its location the previous week, it was categorised as a '+'. Downstream movement was categorised as a '−'. If a fish did not move from its location the previous week (<0.2 km), the data were disregarded. If the data were directional, a run (string of upstream or downstream values) would tend to be long and the number of different runs would tend to be small (Ramsey & Schafer 1997); this characteristic run would signify a fish migration. The null hypothesis was that a radio-tagged fish's movement was nondirectional.

The nonparametric runs test statistic (μ) was expressed as:

$$\mu = \frac{2mp}{m+p} + 1,$$

where m is the number of '−' and p the number of '+'.

The standard deviation of the number of runs (δ) was expressed as:

$$\delta = \frac{\sqrt{2mp(2mp-m-p)}}{\sqrt{(m+p)^2(m+p-1)}}.$$

The test statistic used was:

$$Z = \frac{(\text{number of runs}) - \mu + C}{\delta}$$

where C is a continuity correction. C was set at 0.5 if the number of runs was less than μ and at −0.5 if the number of runs was greater than μ.

A t-test was used to investigate if consecutive-year migrants from 1999 moved upstream in 2000 at the same rate as fish radio tagged in 2000. Weekly movement rate for each fish was considered a sample.

Seven geographic areas were delineated: Salmon River, SFSR, lower EFSFSR (mouth upstream to Johnson Creek), upper EFSFSR (Johnson Creek upstream to headwaters), tributaries to upper EFSFSR, Johnson Creek and tributaries to Johnson Creek (Fig. 1). A chi-square test was used to evaluate if there was a statistical difference ($\alpha = 0.05$) between the observed number of bull trout entering the upper EFSFSR and those entering Johnson Creek.

Results

Migrations and movements

Movements of individual fish and groups of fish were qualitatively assessed to be strongly directional migrations, even though the runs test indicated that individual fish exhibited both nondirectional (43 fish) and directional (12 fish) movements in both years. Analysis of group movements supported the qualitative assessment; fish from both 1999 and 2000 as groups exhibited directional patterns [1999: runs test, $N = 31$, $P = 0.01$ (Fig. 2a); 2000: runs test, $N = 30$, $P = 0.01$ (Fig. 2b)] considered to be migrations.

Fish tagged in 1999 and returning in 2000 migrated upstream rapidly in early July. Consecutive-year migrants moved upstream more quickly during the week of 3 July 2000 than other bull trout initially radio tagged in 2000 (d.f. = 27, $F = 13.35$, $P = 0.0011$).

Observed movements of 62 fish were classified into one of four patterns: consecutive-year fluvial, non-consecutive-year fluvial, adfluvial and down-river stationary. Three other fish were classified as having experienced tag loss or mortality.

Three fish exhibited a fluvial migration pattern with consecutive-year spawning. After initial tagging in the EFSFSR in late June or July, the fish migrated upstream to the mouth of a small tributary in early July, entered the tributary in mid-August, spawned or staged there from late August through mid-September, then rapidly emigrated downstream by mid-September into the SFSR or Salmon River, where they overwintered. This pattern was repeated beginning in late May or early June of the following year (Fig. 3a). The three fish entered the same major tributary (although not

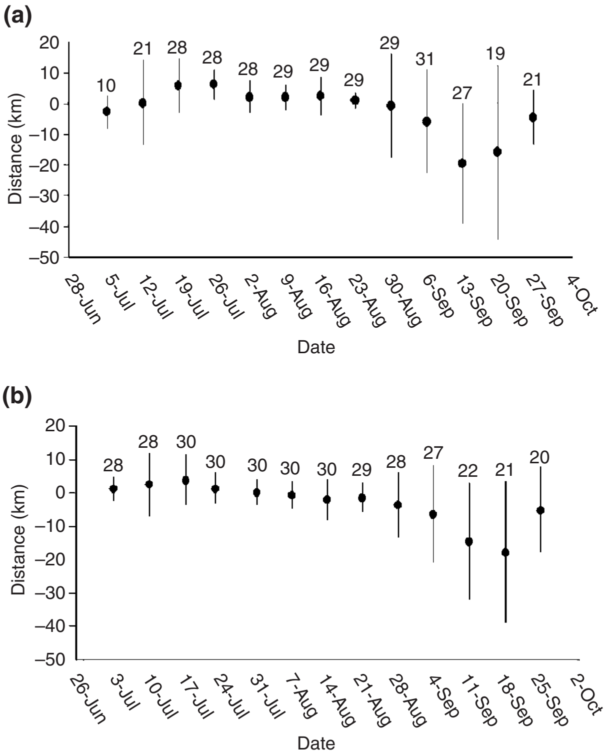

Fig. 2. (a) Bull trout weekly group mean migration rate (km·week^{-1}) from 5 July to 3 October 1999. Positive distances signify an upriver migration and negative distances signify a downstream migration. (b) Bull trout weekly group mean migration rate (km·week^{-1}) from 3 July to 25 September 2000. Positive distances signify an upriver migration and negative distances signify a downstream migration.

necessarily the same smaller, second-order tributary) in 2000 that they had entered in 1999. Two of the three fish maintained their tags into the second winter and migrated downstream to the same area they occupied in 1999.

Fourteen fish exhibited a fluvial migration pattern with nonconsecutive-year spawning. Nine of the 14 fish, which were initially tagged in the EFSFSR in 1999, migrated upstream to the mouth of a small tributary, entered the tributary in early August, spawned or staged there from late August through mid-September, then rapidly emigrated downstream into the SFSR or Salmon River, where they overwintered and remained throughout the summer of 2000 (Fig. 3b). The other five fish, after initial tagging in the EFSFSR in late June or July of either 1999 or 2000, migrated upstream in the EFSFSR but did not enter a small tributary. After the presumed spawning period, these fish moved downstream to overwinter in the SFSR or Salmon River.

For 33 other fish, the migratory pattern was fluvial, although we were unable to determine if they were consecutive-year or nonconsecutive-year spawners. Twenty-one fish initially tagged in the EFSFSR migrated upstream to the mouth of a small tributary, entered the tributary and spawned or staged there in late August through mid-September, after which their radio tags were found on stream-banks or the streambed (Fig. 3c). The remaining 12 fish, which were initially tagged in the EFSFSR in 2000, migrated upstream to the mouth of a small tributary, staged there, entered the tributary in mid-August, spawned or staged there from late August through mid-September, then rapidly emigrated downstream to a large river (SFSR or Salmon River), where they overwintered (Fig. 3d). These fish were not tracked long enough to determine their spawning periodicity.

The adfluvial migration pattern (five fish) consisted of initial tagging in the Glory Hole (or nearby, <3 km downstream in the EFSFSR), downstream movement starting in late June, staging at the mouth of a small tributary in the EFSFSR by mid-August, entrance into the tributary in late August, spawning or staging in the tributary, rapid dispersal out of the tributary stream after spawning, followed by migration upstream in the EFSFSR back to the Glory Hole and residence in the Glory Hole for the winter (Fig. 3e). Only one fish completed this entire migration pattern, but four other fish completed portions of it.

The downriver-stationary pattern (seven fish) consisted of initial tagging in the EFSFSR, movement downstream immediately after tagging into the SFSR or Salmon River and residence there through the remainder of the study (Fig. 3f). Some of these fish moved upriver in the large river habitats but made no observable spawning migrations.

None of the 29 fish tagged in the EFSFSR downstream of the Johnson Creek confluence moved upstream into Johnson Creek during the two summers. Fish moved preferentially into the upper EFSFSR rather than into Johnson Creek (chi-square test, $P = 0.001$).

Prespawning activity

Bull trout first entered a small tributary to EFSFSR on 29 July 1999 and 10 July 2000. One fish that had entered a small tributary on 5 August 1999 was found downstream in the EFSFSR on 10 August 1999, but then had returned to the tributary by September 1. Similar activity was observed with a second fish in 2000. On 15 July, it entered a small tributary, exited the tributary between 27 July and 31 July, and was located again in a still smaller upstream tributary on 29 August.

Groups (five to 25) of large untagged fish (>400 mm) were observed congregating in pools near the mouth of small headwater tributaries from

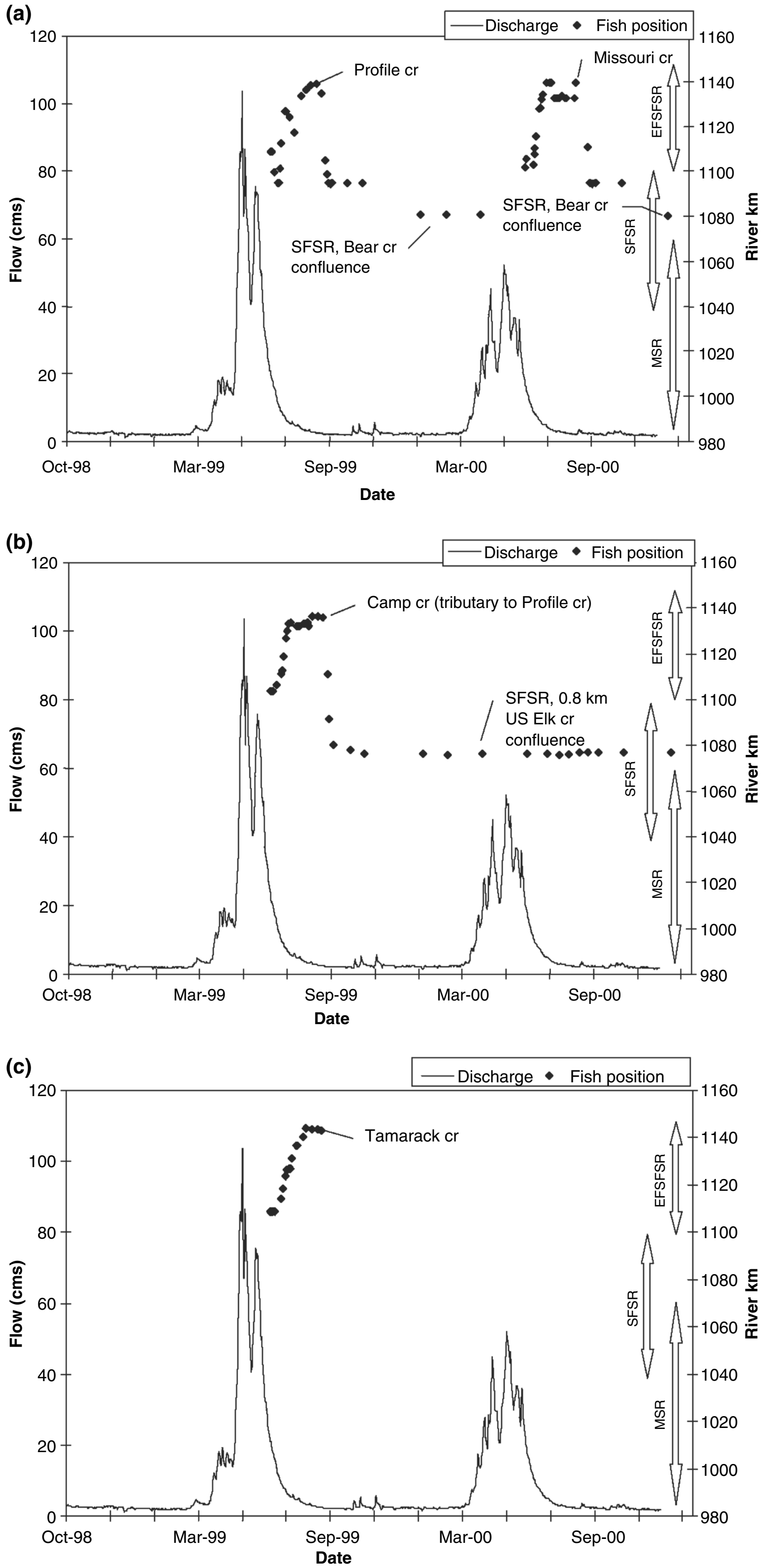

Fig. 3. (a) Bull trout W0076 fluvial migration pattern with consecutive-year spawning observed in the South Fork Salmon River subbasin, 1999–2000. (b) Bull trout W0070 fluvial migration pattern with downriver overwintering and alternate-year spawning observed in the South Fork Salmon River subbasin, 1999 and 2000. (c) Bull trout W0071 fluvial migration pattern (with probable tag expulsion after spawning) observed in the South Fork Salmon River subbasin, 1999–2000. (d) Bull trout O0010 fluvial migration pattern with downriver overwintering observed in the South Fork Salmon River subbasin, 1999–2000. Fish was tagged in 2000. (e) Bull trout W0146 adfluvial migration pattern observed in the South Fork Salmon River subbasin, 1999–2000. (f) Bull trout W0072 fluvial migration pattern (downriver-stationary) in the South Fork Salmon River subbasin, 1999–2000.

mid-July to mid-August. On 7 August 2000, nine radio-tagged fish were in one tributary of the EFSFSR but none of them had paired for spawning. Three days later, the nine fish still had not paired.

Spawning period

No radio-tagged bull trout were observed spawning during this study, so the spawning locations and

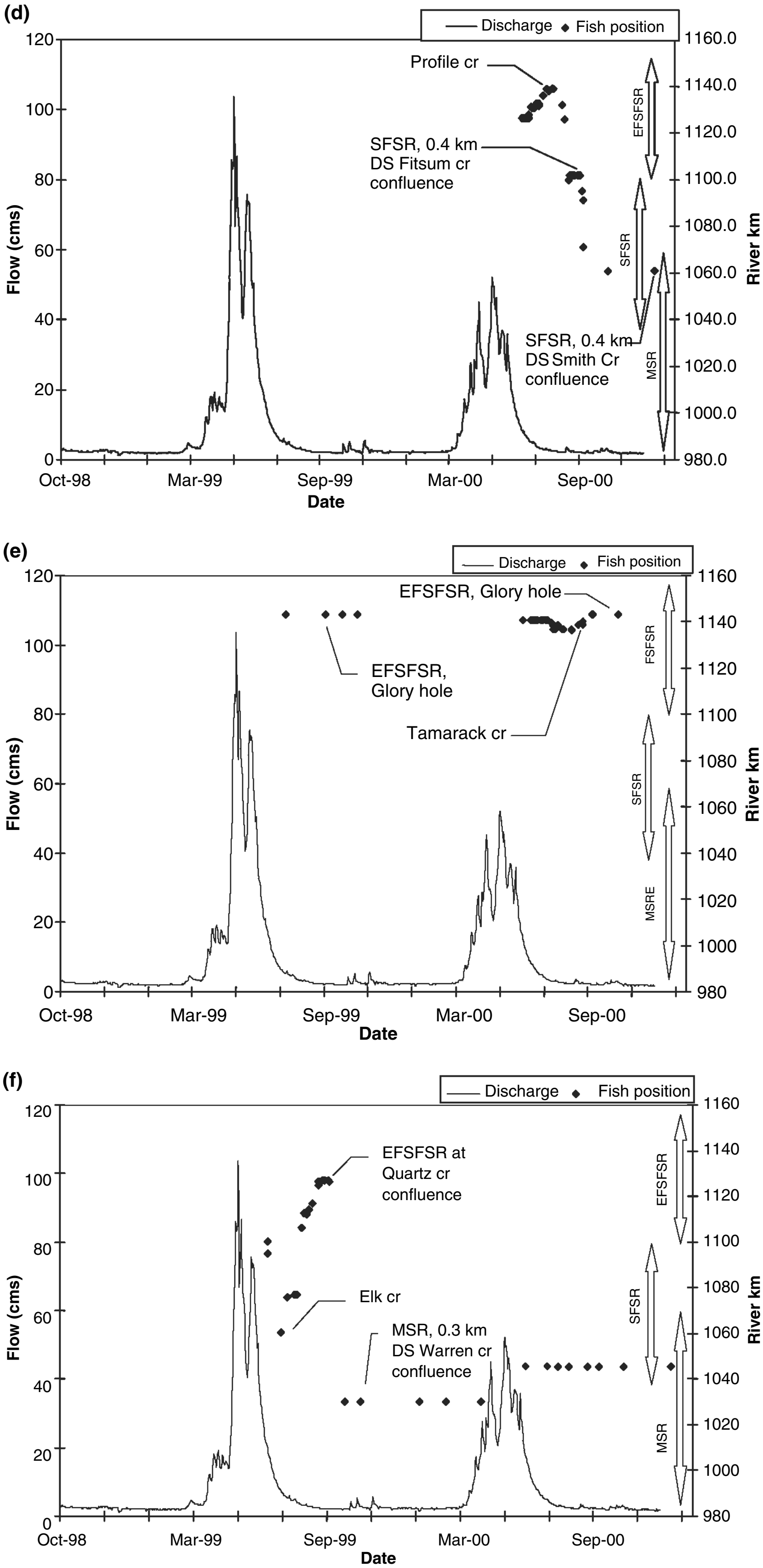

Fig. 3. Continued.

spawning period were inferred by a combination of movements of radio-tagged fish and behavioural observations of untagged fish. Prespawning movements of fish into small tributaries occurred over the period July–September in both 1999 and 2000. Contacts with 44 tagged fish were made in nine small headwater tributaries to the EFSFSR during this period. Two other fish were in the upper EFSFSR in July, August and September, but they did not enter into a tributary. Radio-tagged fish were not distributed in the same tributaries in the same proportions in 1999 and 2000. Four fish tagged in the Glory Hole also migrated downstream and entered into a small tributary during this period.

Pairing, redd construction and spawning of untagged fish were observed from mid-August to mid-September in both years. In 1999, pairing was first observed on 10 August in a tributary. Subsequent observations of pairing in tributaries occurred on 24, 25 and 31 August, and 1, 8 and 14 September. In 2000, untagged fish were first observed spawning on 28 August, with pairing and redd construction observed through 8 September.

Tag loss

Radio-tagged fish experienced high (47%) tag loss in 1999 and 2000 while in the EFSFSR headwater tributaries. In 1999, 27 bull trout entered into smaller tributaries of the EFSFSR and 12 tags (44%) were later recovered either in the tributaries or on the streambanks. In 2000, 24 bull trout entered small tributaries of the EFSFSR and 12 transmitters (50%) were recovered. None of the fish losing their tags lost them during the period soon after tagging. The average time between tagging and tag loss for those fish losing their tags was 70 days (minimum, 28 days; maximum, 351 days).

Postspawning

By 20 September of both 1999 and 2000, radio-tagged bull trout still retaining their radio tags had exited the spawning tributaries. Radio-tagged bull trout migrated downstream rapidly at this time up to 106.4 km in 1 week.

Overwintering

Fluvial bull trout overwintered in the lower SFSR and Salmon River. Of the tagged fish located during the winter, 20 were in the lower SFSR, nine were in the lower Salmon River and six were in the upper Salmon River. Bimonthly tracking during winter detected very little movement. Three fish that overwintered in 1999 migrated upstream in 2000, then returned downstream to overwinter in 2000 at the same locations (SFSR and Salmon River) as in 1999.

Discussion

Migrations and movements

The presence of several variations of fluvial migration patterns (Fig. 3a–d,f) as well as an adfluvial pattern (Fig. 3e) for the bull trout of the EFSFSR is consistent with the great plasticity of life histories and migratory patterns observed in the species elsewhere in the region (Rieman & McIntyre 1993). Except for the downriver-stationary pattern, which is difficult to interpret and may have resulted from ecological factors, stress associated with tagging or a combination of those factors, all the patterns have been described elsewhere in other bull trout stocks.

All three fluvial migrants exhibiting consecutive-year migrations into tributaries entered the same tributaries they had entered the previous year. Other studies have also documented consecutive-year spawning site fidelity (Swanberg 1997; Hvenegaard & Thera 2001). In the Kakwa River, Alberta, Hvenegaard & Thera (2001) reported that 10 of 13 (77%) bull trout displayed spawning site fidelity by returning to a specific tributary in each spawning season.

The nonconsecutive-year spawning pattern observed in this study, in which fish migrated into small tributaries to spawn, subsequently overwintered downriver in a large river and remained in the large river habitat throughout the next year, has been documented in other fluvial bull trout populations (Burrows et al. 2001; Hvenegaard & Thera 2001). In the Kakwa River, Hvenegaard & Thera (2001) reported that 18 of 27 bull trout (67%) tracked through more than two successive spawning seasons displayed a tendency towards alternate-year spawning. In our study, nonspawning consecutive-year migrants did not typically migrate upstream to the mouth of a small tributary, but instead typically remained in the lower EFSFSR or SFSR near the mouth of the Secesh River, only migrating a portion of the distance to the tributaries. In spawning years, spawners would enter small tributaries. The 18-month study period was not long enough to determine whether the pattern was alternate-year spawning, spawning at longer intervals or tag loss during the winter.

Because of the short life of some tags, early expulsion of some tags and the short study duration, we were not always able to determine if the tagged fish were consecutive-year or alternate-year spawners. In 21 cases, for example, a fish swam upstream, staged in or at the mouth of a small tributary, entered and spawned or staged in that tributary and lost its tag soon afterward. Similar high rates of tag loss were also observed by Schill et al. (1994) and Elle et al. (1994) in Rapid River, Idaho. In the McLeod River, Alberta, Carson (2001) reported one of nine radio-tagged bull trout that entered a small tributary to spawn lost its tag either due to predation or tag expulsion. For 12 other fish tagged in 2000, the tag transmissions were adequate to demonstrate a fluvial pattern with overwintering downstream in a larger river, although the spawning periodicity was not determined. Swanberg (1997) observed a similar fluvial pattern in the Blackfoot River, Montana. In Rapid River, Elle et al. (1994) also documented fluvial bull trout that migrated upstream to a tributary, entered the tributary to spawn, emigrated rapidly after spawning and resided in the Salmon River, a large river downstream of the spawning tributary.

The adfluvial pattern observed in this study differed greatly from fluvial patterns in that fish migrated downstream out of the Glory Hole in late June to the mouth of a small tributary, apparently spawned in the small tributary and returned upriver to the Glory Hole during the winter. Adfluvial life histories are common in relict bull trout in the Pacific Northwest (Rieman & McIntyre 1993). The adfluvial pattern in EFSRSR bull trout is similar to migrations described by Fraley & Shepard (1989). In that study, adult bull trout entered tributaries from July through September, spawned during September and early October, exited the tributary after spawning and returned (downstream in this case) to the lake to overwinter. Juveniles emigrated from the tributaries into the Flathead River mainly in June and August and continued downstream until reaching Flathead Lake. Their results differed from our results in that the EFSFSR juveniles would have to swim upstream to mature in the Glory Hole and spawn downstream of their rearing area. As the Glory Hole has only been present for 60 years, this migration pattern would have had to develop over a short time period. Carson (2001) observed two bull trout perform a downstream spawning migration in the McLeod River, Alberta. These two fish were radio tagged in the river where they travelled downstream to the mouth of a tributary, entered the tributary, spawned and returned upstream to the capture site in the river. This study and the downstream migration of fish observed in the EFSFSR further raises questions about the mechanisms and cues used in identifying and locating spawning areas. Power (2002) suggested that 'olfaction, together with habitat familiarity and solar navigation, seem to be the most likely modalities involved' (p. 30) in charr homing.

The diversity of migration patterns (fluvial and adfluvial) observed in EFSFSR bull trout is similar to that observed in other charrs elsewhere (Johnson 1980; Kircheis 1980; Naslund 1990). Nordeng (1961) and McCart (1980) summarised migratory characteristics of arctic charrs (*Salvelinus alpinus*), in their regions and reported that a diversity of life history and migration patterns existed. Although anadromous life histories are common in many locations for arctic charr and are typically preferentially displayed by females over males (Mortensen & Christensen 1983), fluvial and adfluvial life histories also have been reported, especially in landlocked situations (Kircheis 1980; Reist 1989). Charrs have been characterised as having evolved migratory and exploratory life histories as adaptations favouring historical colonisation along glacial margins, areas typically characterised as cold, unproductive and unpredictable. Exploration may be adaptive as an effective colonising mechanism, and migrations may allow charrs access to better food supplies in rivers, lakes or the ocean, wherever opportunities arise (Power 2002). Growth of charrs has been shown in numerous cases to be strongly related to the productivity of the habitat (e.g., Barbour & Einarsson 1987; Rubin 1993) and fecundity (and presumably fitness) positively related to fish size (Johnson 1980). Such historical migratory and exploratory adaptations in charrs may also serve a relict charr such as EFSFSR bull trout well in its habitat. Productivity for fish in the batholith-dominated SFSR basin is low, and spawning areas in the headwaters may provide little food or overwintering habitat. In this situation, a migratory, fluvial life history may be the most adaptive life history, particularly where a lack of large lakes in the upper EFSFSR basin prevents the development of all but a modest adfluvial life history (i.e., associated with the Glory Hole; Fig. 3e).

In this study, three spatial units were identified based on the results of radio tagging: overwintering habitat, migrational corridor and spawning and early rearing habitat (Fig. 1). Overwintering habitat was in the larger rivers including the SFSR and a portion of the Salmon River. The Glory Hole was also identified as overwintering habitat for the adfluvial fish (Fig. 1). Migration corridors consisted of segments of the SFSR and the EFSFSR from its confluence with the SFSR (river km 1099.9) upstream to the Glory Hole (river km 1143.0) including Johnson Creek. Spawning and rearing habitat was in the small tributaries including several small tributaries to EFSFSR as well as headwater tributaries to those tributaries (Fig. 1). Although feeding studies were not conducted, larger migratory adults may feed opportunistically in all these habitats.

The entry of 29 radio-tagged bull trout into the upper EFSFSR, but none into Johnson Creek, indicated that the tagged fluvial bull trout preferentially selected the upper EFSFSR over Johnson Creek (Fig. 3). The reason for the large difference in numbers of tagged fish entering the two rivers is not clear. Johnson Creek is known to contain bull trout. The Nez Perce Tribe operated an upstream passage weir (pickets 4 cm apart) from 26 June through 13 September 2000, and during that period, 17 bull trout (390–510 mm TL) were collected in the trap and placed upstream of the weir (M. Daniel, Nez Perce Tribe, McCall, Idaho, personal communication). In addition, the Nez Perce Tribe also had a downstream migrant screw-trap located on lower Johnson Creek, where they captured 55 bull trout; four of them were greater than 350 mm FL. It is possible that our sample of radio-tagged fish did not adequately include Johnson Creek fluvial bull trout. It is also possible that the migration timing of Johnson Creek fish did not coincide with when we radio tagged fish. It may also be that the fluvial bull trout population in Johnson Creek and its tributaries is small relative to that in the upper EFSFSR and tributaries.

Spawning period

The bull trout spawning observed in late August through mid September in 1999 and 2000 in tributaries was associated with maximum water temperatures of 7.4–12.8 °C. A drop in water temperature from 12 to 9–10 °C occurred in most tributaries during early September. The observed spawning time and water temperatures were similar to those observed for resident bull trout by Adams (1994) in tributaries to the Weiser River, Idaho. In the Rapid River, Idaho, Schill et al. (1994) observed fluvial bull trout spawning in late August through mid-September as water temperatures dropped from 10 to 6.5 °C. Fraley & Shepard (1989) observed adfluvial bull trout spawning when water temperatures dropped below 9 °C.

The three radio-tagged bull trout that were consecutive-year spawners returned to the same tributaries as the previous year. Swanberg (1997) observed similar consecutive-year spawning in a tributary of the Blackfoot River, Montana.

Overwintering

The observation that radio-tagged fluvial bull trout moved little during the winter months (less than 1 km) is similar to results reported elsewhere. Swanberg (1997) reported that movements during the winter were very local, never exceeding 300 m. Elle et al. (1994) also found that fluvial bull trout from the Rapid River, Idaho typically remained in one habitat unit for the overwintering period and generally moved less than 100 m between contacts. Even with the reduced tracking schedule in winter, the evidence indicates that overwintering movements were much less extensive than in other seasons.

While observing the locations of the radio-tagged bull trout in the winter, we observed the fish using large deep pools and runs and avoiding shallow riffles. This habitat use is similar to that reported by Schill et al. (1994) and Elle et al. (1994) in the Salmon River near Riggins, Idaho. In our study, three radio-tagged fluvial bull trout exhibited site fidelity to the overwintering habitat by returning to the same location as the previous year. Swanberg (1997) also observed overwintering site fidelity in three radio-tagged bull trout in the Blackfoot River, Montana. Little overlap of EFSFSR fluvial bull trout overwintering habitat was observed with the Rapid River fluvial bull trout overwintering locations (Elle et al. 1994; Schill et al. 1994), even though the two groups had free access to the same overwintering sites. Of the 63 fish tagged and subsequently contacted in overwintering habitats [38 from this study, 17 from Elle et al. 1994 and eight fish from Schill et al. (1994)], only one fish from our study and one fish from Schill et al. (1994) used the same overwintering locations. As the overwintering and spawning areas of the radio-tagged bull trout from the Rapid River and the SFSR do not overlap, it appears that they are separate stocks with not only distinct spawning areas, but also distinct overwintering areas.

Management significance of diverse migration patterns

Results of this study show that bull trout in the EFSFSR have evolved a variety of migrational patterns, both fluvial and adfluvial, similar to bull trout in the region and other charr species throughout the northern hemisphere. Fish from the EFSFSR also exhibited migration patterns consistent with the idea that several distinct stocks of bull trout exist in the SFSR basin. The highly variable life histories and migration patterns associated with this species is consistent with genetic results (Taylor et al. 1999) indicating that most of the molecular genetic variations occur at the interpopulation and inter-region (coastal vs. interior) levels. Maintenance of bull trout habitat for such a highly migratory species composed of numerous stocks will thus require actions in a variety of habitats, from the larger main stem rivers to the smallest tributaries.

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References

Adams, S.B. 1994. Bull trout distribution and habitat use in the Weiser River drainage, Idaho. Master of Science Thesis. Moscow, ID: University of Idaho.

Baker, R.R. 1978. The evolutionary ecology of animal migration. New York: Holmes and Meier Publishers.

Barbour, S.E. & Einarsson, S.M. 1987. Ageing and growth of charr, *Salvelinus alpinus* (L.), from three habitat types in Scotland. Aquaculture and Fisheries Management 18: 63–72.

Berg, O.K. & Berg, M. 1993. Duration of sea and freshwater residence of arctic char (*Salvelinus alpinus*), from the Vardnes River in Northern Norway. Aquaculture 110: 129–140.

Burrows, J., Euchner, T. & Baccante, N. 2001. Bull trout movement patterns: Halfway River and Peace River progress. In: Brewin, M.K., Paul, A.J. & Monita, M., eds. Bull trout II conference proceedings. Calgary, AB: Trout Unlimited Canada, pp. 153–157.

Carson, R.J. 2001. Bull trout spawning movements and homing behavior back to pre-spawning locations in the McLeod River, Alberta. In: Brewin, M.K., Paul, A.J. & Monita, M., eds. Bull trout II conference proceedings. Calgary, AB: Trout Unlimited Canada, pp. 137–140.

Clayton, J.L. & Megahan, W.F. 1997. Natural erosion rates and their prediction in the Idaho batholith. Journal of the American Water Resources Association 33: 689–703.

Doucett, R.R., Power, M., Power, G., Caron, F. & Reist, J.D. 1999. Evidence for anadromy in a southern relict population of Arctic charr from North America. Journal of Fish Biology 55: 84–93.

Elle, S., Thurow, R. & Lamansky, T. 1994. Rivers and streams investigations. Boise, ID: Idaho Department of Fish and Game. Job Performance Report Project F-73-R-16, Subproject II, Study IV.

Fraley, J.J. & Shepard, B.B. 1989. Life history, ecology and population status of migratory bull trout (*Salvelinus confluentus*) in the Flathead Lake and River System, Montana. Northwest Science 63: 133–143.

Gulseth, O.A. & Nilssen, K.J. 2001. Life-history traits of charr, *Salvelinus alpinus*, from a high Arctic watercourse on Svalbard. Arctic 54: 1–11.

Hogen, D.M. 2002. Spatial and temporal distribution of bull trout (*Salvelinus confluentus*) in the upper East Fork South Fork Salmon River watershed, Idaho. Master of Science Thesis. Moscow, ID: University of Idaho.

Hvenegaard, P.J. & Thera, T.M. 2001. Monitoring the bull trout (*Salvelinus confluentus*) spawning run in Lynx Creek, a tributary to the Kakwa River, West Central Alberta. In: Brewin, M.K., Paul, A.J. & Monita, M., eds. Bull trout II conference proceedings. Calgary, AB: Trout Unlimited Canada, pp. 153–157.

Johnson, L. 1980. The arctic charr, *Salvelinus alpinus*. In: Balon, E.K., ed. Charrs. Salmonid fishes of the genus *Salvelinus*. Hague, the Netherlands: D. W. Junk, pp. 15–98.

Johnson, L. 1989. The anadromous arctic charr, *Salvelinus alpinus*, of Nauyuk Lake, N. W. T., Alaska. Physiology and Ecology Japan 1: 201–227.

Kircheis, F.W. 1980. The landlocked charrs of Maine: the sunapee and the blueback. In: Balon, E.K., ed. Charrs. Salmonid fishes of the genus *Salvelinus*. Hague, the Netherlands: D. W. Junk, pp. 749–763.

Klamt, R.R. 1976. The effects of coarse granitic sediment on the distribution and abundance of salmonids in the central Idaho batholith. Master of Science Thesis. Moscow, ID: University of Idaho.

Kuzis, K. 1997. Watershed analysis of the upper East Fork South Fork Salmon River, Volumes 1 & 2. Report of KK Consulting to US Forest Service, McCall, ID, USA.

McCart, P.J. 1980. A review of the systematics and ecology of arctic char, *Salvelinus alpinus*, in the western Arctic. Canadian Technical Report of Fisheries and Aquatic Sciences 935, Ottawa, Ontario, Canada.

Mortensen, E. & Christensen, B. 1983. Population dynamics of arctic charr, *Salvelinus alpinus* (L.) in River Narssaq, Greenland. In: Hammar, J. & Nyman, L., eds. Proceedings of the second ISACF workshop on arctic char, 1982. Drottningholm, Sweden: Institute of Freshwater Research, pp. 107–125.

Naslund, I. 1990. The development of regular seasonal habitat shifts in a landlocked arctic charr, *Salvelinus alpinus*, population. Journal of Fish Biology 36: 401–414.

Naslund, I. 1992. Upstream migratory behaviour in landlocked arctic charr. Environmental Biology of Fishes 33: 265–274.

Nordeng, H. 1961. On the biology of char (*Salmo alpinus* L.) in Salangen, North Norway. I. Age and spawning frequency determined from scales and otoliths. Nytt Magasin for Zoologi 10: 67–123.

Power, G. 2002. Charrs, glaciations, and seasonal ice. Environmental Biology of Fishes 64: 17–35.

Ramsey, F.L. & Schafer, D.W. 1997. The statistical sleuth: a course in methods of data analysis. Belmont, CA: Wadsworth Publishing Company.

Reist, J.D. 1989. Genetic structure of allopatric populations and sympatric life history types of charr, *Salvelinus alpinus/malma*, in the western Arctic, Canada. Physiology and Ecology Japan 1: 405–420.

Rieman, B.E. & McIntyre, J.D. 1993. Demographic and habitat requirements for conservation of bull trout. Boise, ID: US Department of Agriculture, Forest Service, Intermountain Research Station. Gen. Tech. Rep. INT-302.

Rubin, J. 1993. The exceptional growth of the arctic charr, *Salvelinus alpinus* (L.) in Lake Geneva. Aquatic Sciences 55: 76–86.

Schill, D., Thurow, R. & Kline, P. 1994. Seasonal movement and spawning mortality of fluvial bull trout in Rapid River, Idaho. Boise, ID: Idaho Department of Fish and Game. Project: F-73-R-15, Job 2.

Shepard, B., Pratt, K. & Graham, P. 1984. Life histories of westslope cutthroat and bull trout in the Upper Flathead River Basin, Montana. Kalispell, MT: Montana Department of Fish, Wildlife and Parks.

Stelfox, J.D. 1997. Seasonal movements, growth, survival, and population status of the adfluvial bull trout population in lower Kananaskis Lake, Alberta. In: Mackay, W.C., Brewin, M.K. & Monita, M., eds. Friends of the bull trout conference

proceedings. Calgary, Alberta, Canada: Bull Trout Task Force (Alberta), c/o Trout Unlimited, pp. 309–316.
Stowell, R., P. Howell, Rieman, B.E. & McIntyre, J. 1996. An assessment of the conservation needs for bull trout. Missoula, MT: US Department of Agriculture, Forest Service, Northern Region, Report R1-96-71.
Swanberg, T.R. 1997. Movements of and habitat use by fluvial bull trout in the Blackfoot River, Montana. Transactions of the American Fisheries Society 126: 735–746.
Taylor, E.B., Pollard, S. & Louie, D. 1999. Mitochondrial DNA variation in bull trout (*Salvelinus confluentus*) from northwestern North America: implications for zoogeography and conservation. Molecular Ecology 8: 1155–1170.
US Fish and Wildlife Service (USFWS) 1998. Endangered and threatened wildlife and plants; determination of threatened status for the Klamath River and Columbia River distinct population segments of bull trout. Final rule. Federal Register 63: 31647–31674.
Winter, J.D. 1996. Advances in underwater biotelemetry. In: Murphy, B.R. & Willis, D.W., eds. Fisheries techniques, 2nd edn. Bethesda, MD: American Fisheries Society, pp. 555–590.

Effect of Three Hatchery Lighting Schemes on Indices of Smoltification in Chinook Salmon

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Effect of Three Hatchery Lighting Schemes on Indices of Smoltification in Chinook Salmon

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Abstract.—This experiment examined whether photoperiodic changes induce chinook salmon *Oncorhynchus tshawytscha* to alter various indices of smoltification. The effect of 24 h light, a constant photoperiod of 9 h light : 15 h dark (9L:15D), and a naturally increasing photoperiod on plasma levels of thyroxine and cortisol, hematocrit, condition factor (K), and hepatosomatic index (HSI) was tested during the period of smoltification in chinook salmon. The 24-h-light group grew faster than the other two groups, but was significantly larger than one of the two groups on only three sampling dates. Mean plasma thyroxine and cortisol levels were highest in the natural-photoperiod group and lowest in the 9L:15D group. Mean plasma cortisol levels increased significantly in the increasing and 9L:15D photoperiod groups but were unchanged in the continuous-light group. Mean HSI decreased faster in the natural-photoperiod group than in the other two groups. Mean hematocrit did not change in the natural-photoperiod group but decreased in the other two groups. Mean K decreased in the natural-photoperiod and 24-h-light groups, but not in the 9L:15D group. The natural-photoperiod group clearly showed a more coordinated and complete smoltification based on the indices measured: highest mean and peak levels of both thyroxine and cortisol, greatest decrease in HSI, greatest decrease in K (along with the 24-h-light group), and least decrease in hematocrit. These results show that a natural photoperiod is beneficial and that continuous light or a short, unchanging photoperiod are detrimental to smoltification in chinook salmon. Hatcheries should account for photoperiodic effects when raising chinook salmon, particularly indoors.

Salmon *Oncorhynchus* spp. and *Salmo* spp. in hatcheries are commonly raised under indoor conditions of a short day and unchanging photoperiod or 24 h of light. A short day and unchanging photoperiod, in which the lights are turned on and off with the arrival and departure of the hatchery staff, is often used for fiscal reasons. Continuous (24-h) light is commonly used to improve growth of the fish (Clarke et al. 1978; McCormick et al. 1987; Saunders et al. 1989). Rottiers (1992) found that Atlantic salmon *S. salar* raised under 16 h light : 8 h dark (16L:8D) grew faster than those under 8L:16D. However, these nonchanging lighting schemes may deprive fish of important cues for smoltification. This deprivation may detrimentally affect survival and imprinting of the juvenile salmon and, thus, return rates of adult salmon. Wedemeyer et al. (1980) concluded that improvements in hatchery practices, including proper regulation of photoperiod, will help increase survival and return rates.

Changes in photoperiod are the most predictable events which can be used to precisely distinguish the time of year. Higgins (1985) concluded that photoperiod was the primary environmental influence on the induction of smoltification because it was not as variable as temperature changes from year to year. Wagner (1974) found photoperiod to be more important than temperature in determining the onset of smoltification in steelhead (anadromous rainbow trout *O. mykiss*). Baggerman (1963) found that the change in salinity preference in coho salmon *O. kisutch* was photoperiodically controlled and was preceded by an increase in thyroxine levels. Saunders and Henderson (1970) found that Atlantic salmon subjected to a constant 13L:11D or increasing photoperiod smoltified and grew rapidly after entering seawater but that those subjected to a decreasing photoperiod did not smoltify and fared poorly in seawater. Zaugg and Wagner (1973) found that accelerated photoperiodic changes (advancing the onset of long days) resulted in early smoltification and that decelerated photoperiodic changes caused delayed smoltification. Clarke et al. (1985) were able to advance the seawater adaptability of Atlantic salmon by advancing photoperiod, and Duston and Knox (1992) were able to cause Atlantic salmon to acclimate to seawater during the autumn by manipulating photoperiod. Grau et al. (1982) concluded that photoperiod synchronizes the developmental

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changes associated with smoltification. Duston and Saunders (1990) supported the hypothesis that photoperiodic changes ''entrain an endogenous circannual rhythm involved in controlling the completion of smoltification.'' Plasma melatonin levels in Atlantic salmon increase in relation to the duration of the daily dark period (N. R. Bromage and others, Institute of Aquaculture, University of Stirling, personal communication). Therefore, it appears that salmon respond to the increasing photoperiod during spring to initiate the smoltification process.

Increases in hematocrit (Stefansson et al. 1989), plasma thyroxine (Folmar and Dickhoff 1981), and plasma cortisol (Specker and Schreck 1982) and a decrease in condition factor (*K*; Hoar 1939) are known indices of smoltification in salmon. Hepatosomatic index (HSI) also decreases with smoltification as hepatic lipid, glucose, and glycogen levels decrease (Sheridan et al. 1985). Our experiment examines whether chinook salmon *O. tshawytscha* are sufficiently sensitive to photoperiodic changes to alter these indices of smoltification. The null hypothesis being tested is that continuous light; a short, unchanging photoperiod; or an increasing day length photoperiod are inhibitory to smoltification-related increases in hematocrit and plasma thyroxine and cortisol levels and decreases in HSI and *K* during smoltification of chinook salmon.

Methods

The fall chinook salmon stock sampled in this study originated from the Toutle River, Washington, via the Little Manistee River, Michigan. They were raised at McNenny State Fish Hatchery, Spearfish, South Dakota, under natural lighting and photoperiod. On 10 March 1991, approximately 1,200 age-0 chinook salmon were transferred into 90.8-L (71 cm in diameter × 40 cm deep) circular tanks, each assigned to one of three photoperiod groups (approximately 400 fish/group): 24-h light, 9L:15D photoperiod, and a naturally increasing day length photoperiod. In the latter group, the light portion of the photoperiodic cycle was increased 0.5 h/week and ranged from 10L:14D on 17 March to 15.5L:8.5D on 3 June. Light was supplied by broad-spectrum fluorescent lights 40 cm above the water surface. Light intensity at the water surface was 55 lx. Natural light was excluded by black plastic surrounding each tank. The fish were fed Nelson's Sterling Cup Salmon Feed at 2.25% of body weight daily. Water at McNenny Hatchery comes from a well at a constant 11.2°C, and flow into the tanks was 18.9 L/min.

Blood samples were collected from 30 fish/treatment group (90 fish/sampling date) at approximate 1-week intervals from 17 March through 3 June 1991. Ten fish at a time were collected from a tank and placed in a 19-L bucket with a buffered solution of MS-222 (tricaine methanesulfonate) anesthetic (75 mg/L). Individual fish were taken from the bucket and weighed (g), and total length (mm) was measured. Blood was collected from the severed caudal peduncle into heparinized capillary tubes. The liver was then removed and weighed (g). Sampling of all 10 fish took 10 min or less. Blood samples were centrifuged, hematocrit was measured, and plasma was separated and stored at −20°C until analysis.

Plasma thyroxine and cortisol concentrations were assayed with solid-phase radioimmunoassay kits (Micromedic Systems, Horsham, Pennsylvania). The effectiveness of the use of these kits to determine hormone levels in fish has been previously reported (Hoffnagle and Fivizzani 1990). All samples were analyzed in a single assay to eliminate the problem of interassay variability. Standards were prepared by adding known quantities of thyroxine or cortisol to chinook salmon plasma previously charcoal-stripped of all endogenous hormones. Duplicate 5-μL undiluted samples and standards were assayed and the bound fraction counted with a Beckman 5500 gamma counter. Estimates of sample plasma thyroxine or cortisol content were determined from a linear regression between appropriate adjacent points on the standard curve. Due to the limited plasma volume from each fish sampled and predominance of plasma thyroxine analysis in previous studies, plasma triiodothyronine was not assayed in this study.

Condition factor was calculated by the equation: $K = 100 \cdot (\text{body weight, g})/(\text{length, cm})^3$. Hepatosomatic index was calculated by the equation: $\text{HSI} = 100 \cdot (\text{liver weight, g})/(\text{body weight, g})$.

Mean plasma thyroxine and cortisol levels and mean *K*, HSI, and hematocrit for each experimental group were compared among treatment groups and sampling dates by analysis of variance (ANOVA) and the Student–Newman–Keuls multiple-range test. Regressions of mean total length, body weight, HSI, hematocrit, and *K* versus day of the year were used to determine the rate of change in these smoltification indices over time. Dummy variables were used to test for differences in slope of regression lines among groups (Kleinbaum and

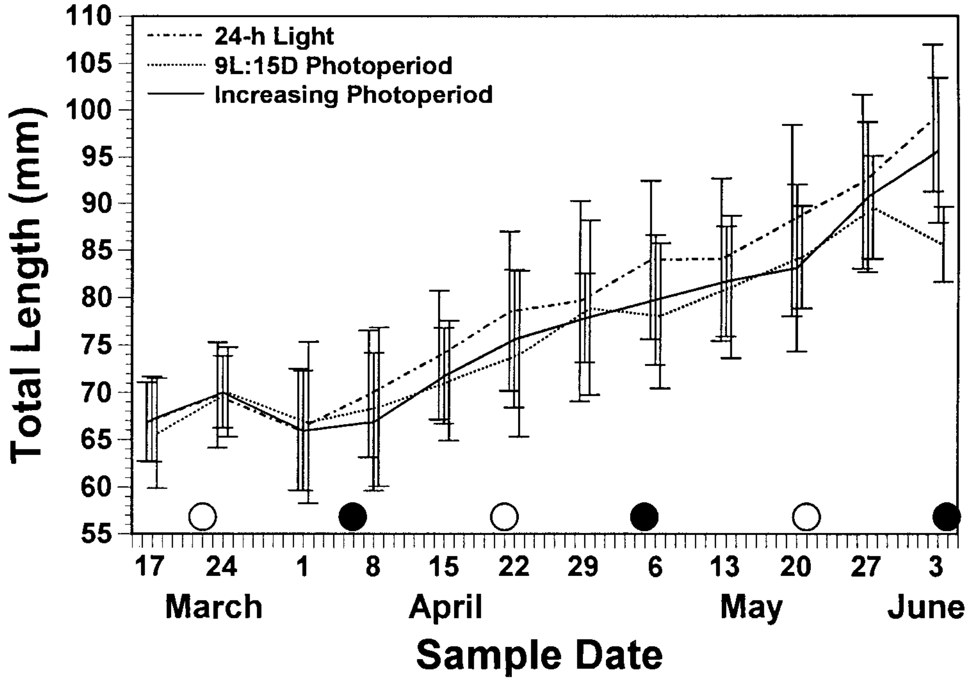

FIGURE 1.—Changes in mean (±SE) length during smoltification of chinook salmon exposed to 24 h light, a constant 9L:15D (9-h-light : 15-h-dark) photoperiod, or a naturally increasing photoperiod. New and full moon dates are indicated by filled and open circles, respectively.

Kupper 1978). All statistical tests were considered to be significant at $\alpha = 0.05$.

Results

The results of this experiment demonstrate differences in the effects of continuous light; a short, unchanging photoperiod; and a naturally increasing photoperiod on smoltification in chinook salmon. The 24-h-light group grew at a faster rate in both length (Figure 1) and weight (Figure 2) than the other two groups ($P \leq 0.0052$). The patterns of change of plasma thyroxine (Figure 3) and cortisol (Figure 4) were similar in each experimental group; however, the natural-photoperiod group had higher peak levels of both hormones ($P = 0.0001$). Mean HSI decreased significantly in all groups ($P \leq 0.0023$; Figure 5). Mean hematocrit significantly decreased in the 24-h-light and 9L:15D groups ($P \leq 0.0038$) but not in the natural-photoperiod group ($P = 0.3132$; Figure 6). Mean K decreased significantly in the natural-photoperiod and 24-h-light groups ($P \leq 0.0255$) but not in the 9L:15D group ($P = 0.1313$; Figure 7).

Growth

The 24-h-light group (0.403 mm/d; $r^2 = 0.9434$) grew significantly faster ($P \leq 0.0029$) in total length than the 9L:15D (0.299 mm/d; $r^2 = 0.9213$) or natural-photoperiod groups (0.355 mm/d; $r^2 = 0.8982$), which were not significantly different from each other ($P = 0.1428$; Figure 1). However, the 24-h-light group was significantly larger than both other groups only on 6 May ($P = 0.0109$) and was also larger than the 9L:15D group on 3 June ($P = 0.0001$). There were no other significant differences among experimental groups on any other sampling date ($P \geq 0.0548$).

The 24-h-light group (0.063 g/d; $r^2 = 0.9204$) grew significantly faster ($P \leq 0.0052$) in body weight than the 9L:15D (0.043 g/d; $r^2 = 0.8515$) or natural-photoperiod groups (0.054 g/d; $r^2 = 0.8982$), which were not significantly different from each other ($P = 0.0704$; Figure 2). However, the 24-h-light group was significantly larger than both other groups only on 6 May ($P = 0.0178$) and was also larger than the natural-photoperiod group on 20 May ($P = 0.0478$) and the 9L:15D group on 3 June ($P = 0.0002$). There were no other significant differences among experimental groups on any other sampling date ($P \geq 0.1093$).

Plasma Thyroxine

Mean plasma thyroxine levels peaked on 29 April for all groups and on 27 May for both the

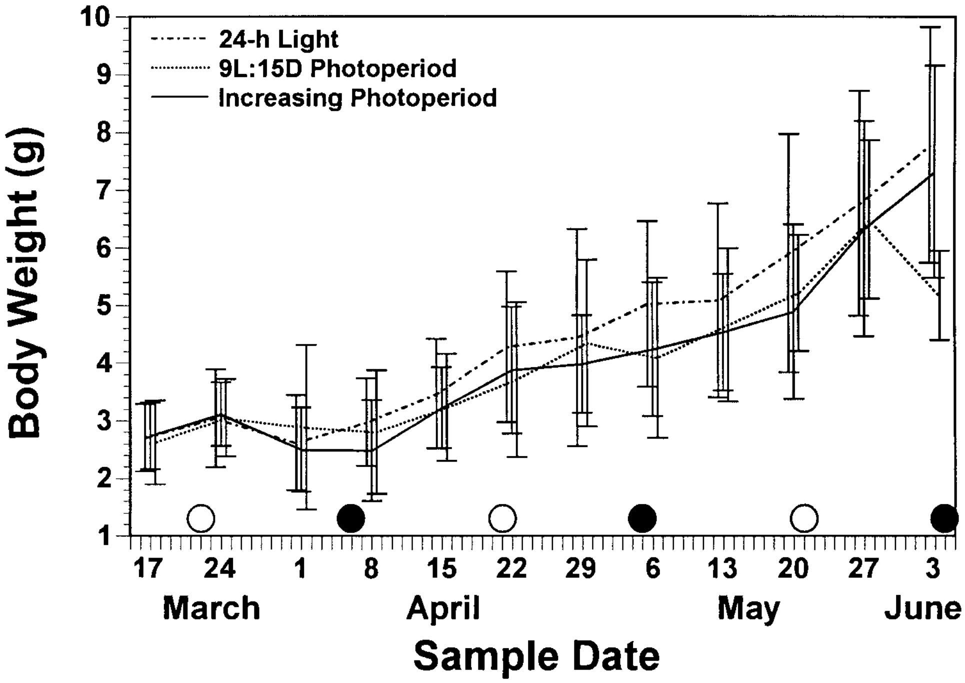

FIGURE 2.—Changes in mean (±SE) weight during smoltification of chinook salmon exposed to 24 h light, a constant 9L:15D photoperiod, or a naturally increasing photoperiod. New and full moon dates are indicated by filled and open circles, respectively.

natural-photoperiod and 24-h-light groups (Figure 3). Mean plasma thyroxine levels were similar for all groups on 29 April ($P = 0.2301$). However, mean plasma thyroxine levels for the simulated natural-photoperiod group were significantly higher ($P = 0.0001$) than those of the other groups on 27 May (27.33 ng/mL). Mean plasma thyroxine levels for the simulated natural-photoperiod group were also significantly higher than those of the other groups on 20 May and 3 June ($P \leq 0.0002$). Mean plasma thyroxine levels for the 24-h-light group peaked on 29 April (17.58 ng/mL) and 27 May (19.87 ng/mL). Mean plasma thyroxine levels for the 9L:15D group were the lowest of the three experimental groups on 7 of the 12 sampling dates and were significantly lower ($P \leq 0.0130$) than one or both other groups on five dates. The 9L:15D group showed only one significant peak (17.99 ng/mL on 29 April) in mean plasma thyroxine level.

Plasma Cortisol

As was seen for plasma thyroxine, plasma cortisol levels peaked on 27 May, when mean levels for the natural-photoperiod and 9L:15D groups reached their highest levels of the experiment (Figure 4). Plasma cortisol levels in the natural-photoperiod group showed little change in the early samples and were consistently the intermediate value of the three experimental groups. However, on 27 May mean plasma cortisol in the natural-photoperiod group peaked at 108.94 ng/mL, which was significantly higher than the other two groups ($P = 0.0001$). Plasma cortisol levels in the 9L:15D group were unchanged and consistently lower than both of the other groups for 9 of the first 10 sampling dates and were statistically lower than one or both of the other groups on 8 of these dates ($P \leq 0.0463$). However, on 27 May mean plasma cortisol levels for the 9L:15D group peaked at 78.13 ng/mL, significantly higher than the 24-h-light group but significantly lower than the natural-photoperiod group ($P = 0.0001$). The ANOVA for mean plasma cortisol levels in the 24-h-light group was significant ($P = 0.0010$), but the multiple comparisons found no significant differences among sampling dates. However, mean plasma cortisol levels in the 24-h-light group were the highest of the three groups on the first eight sampling dates and were significantly higher than one or both other groups on five of these eight dates ($P \leq 0.0463$). On 22 and 29 April mean plasma

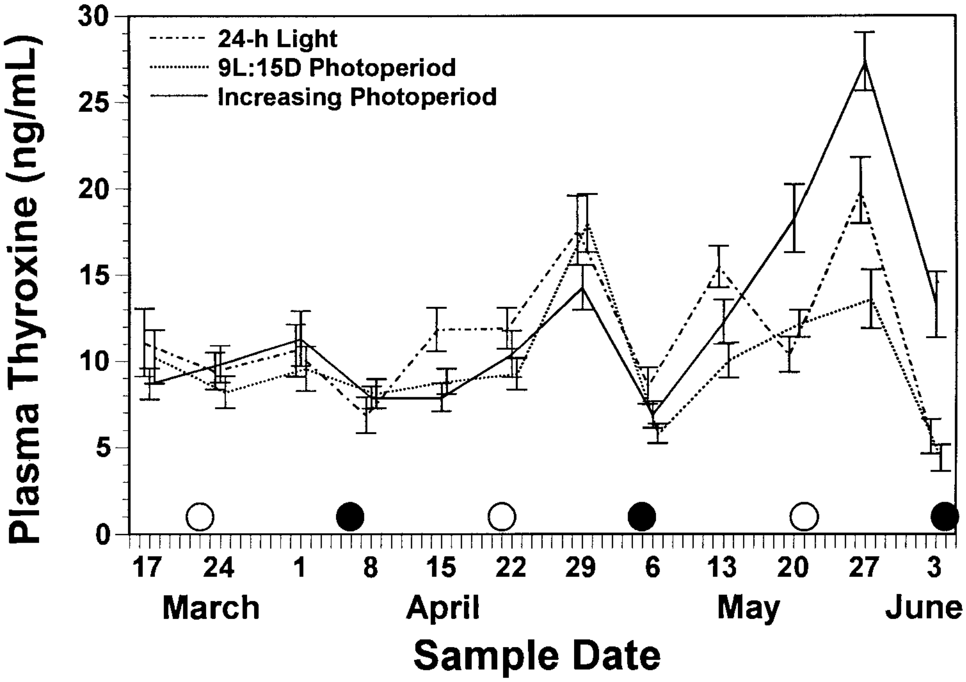

FIGURE 3.—Changes in mean (±SE) plasma thyroxine (ng/mL) during smoltification in chinook salmon exposed to 24 h light, a constant 9L:15D photoperiod, or a naturally increasing photoperiod. New and full moon dates are indicated by filled and open circles, respectively.

cortisol levels in the 24-h-light group were significantly higher than those of the natural-photoperiod group, while on 27 May they were significantly lower than both other groups.

Hepatosomatic Index

Hepatosomatic index decreased significantly over the duration of the experiment in all groups (Figure 5). The rate of decrease in HSI of the natural-photoperiod group (slope = -0.0069, r^2 = 0.8440) was statistically greater ($P \leq 0.00987$) and nearly 50% steeper than that of the other two groups. The rate of decrease in HSI (slope = -0.0042, r^2 = 0.6222) of the 9L:15D group and the 24-h-light group (slope = -0.0043, r^2 = 0.7491) were not statistically different (P = 0.7799).

Hematocrit

Mean hematocrit decreased in both the 24-h-light and 9L:15D groups but did not change over the course of the experiment in the natural-photoperiod group (Figure 6). The rate of decrease in hematocrit (slope = -0.0713, r^2 = 0.5970) of the 24-h-light group was not statistically different from that of the 9L:15D group (P = 0.84193). However, the mean hematocrit for the 24-h-light group was significantly higher than those of the other two groups (P = 0.0005).

Condition Factor

Mean K decreased in both the natural-photoperiod and 24-h-light groups but not in the 9L:15D group (Figure 7). The regression line for mean K was not statistically different (P = 0.5586) between the 24-h-light (slope = -0.001, r^2 = 0.7939) and natural-photoperiod groups (slope = -0.0008, r^2 = 0.4075). Condition factor did not significantly decrease in the 9L:15D fish (P = 0.1313). Condition factor for this group was highest on 10 of the 12 sampling dates (including the final 8), was significantly higher than both of the other groups on two of those dates (20 and 27 May) and over the entire experiment (P = 0.0001), and was not significantly lower than any other experimental group on any sampling date.

Discussion

The results of this experiment show that constant light and a short, unchanging photoperiod had deleterious effects on the measured smoltification indices in these chinook salmon. Although

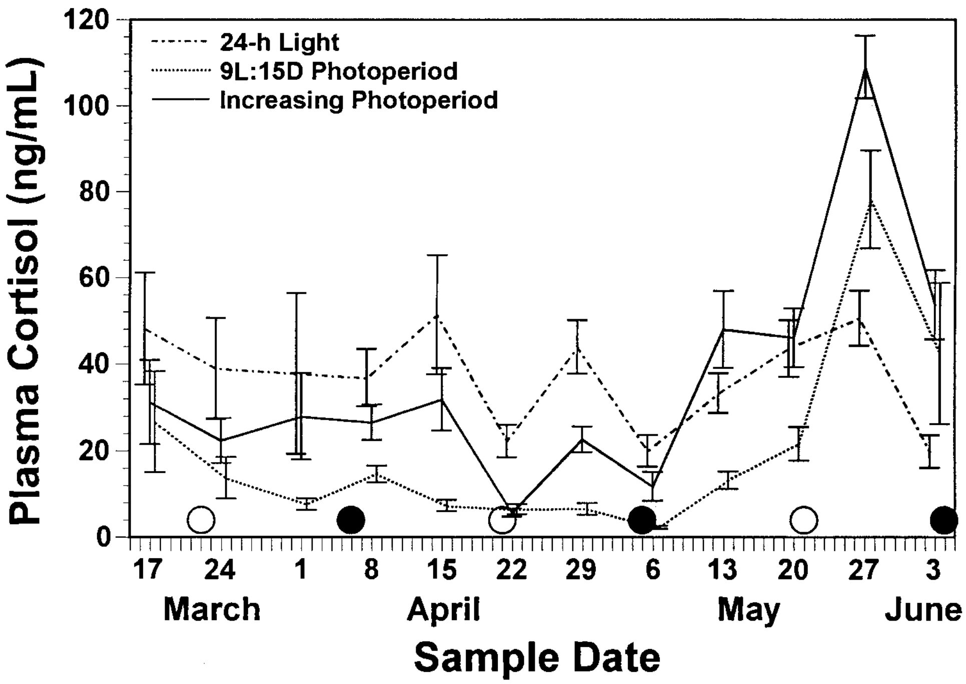

FIGURE 4.—Changes in mean (±SE) plasma cortisol (ng/mL) during smoltification in chinook salmon exposed to 24 h light, a constant 9L:15D photoperiod, or a naturally increasing photoperiod. New and full moon dates are indicated by filled and open circles, respectively.

the 24-h-light group grew faster than the other two groups, and each experimental group showed signs of undergoing smoltification in at least some of the five indices measured, the natural-photoperiod group had the level or rate of change of variables most indicative of smoltification: highest mean and peak levels of both thyroxine and cortisol, greatest decrease in HSI, greatest decrease in *K* (along with the 24-h-light group) and least decrease in hematocrit. The natural-photoperiod group clearly showed a more coordinated and complete smoltification based on the indices measured.

Growth

The 24-h-light group grew at a significantly greater rate in both total length and body weight, although mean size of sampled fish rarely differed on any sampling date. These results are similar to previous studies reporting that Atlantic salmon reared under 24-h-light, a 20L:4D, or 16L:8D photoperiod grew faster than those reared under natural or constant 12L:12D or shorter day length photoperiods (Lundqvist 1980; Saunders and Henderson 1988; Saunders et al. 1989). McCormick et al. (1987) found a reduction in growth rate for Atlantic salmon when transferred from 24 h light to a natural photoperiod. Stefansson et al. (1990) found that continuous light improved growth of Atlantic salmon over 16L:8D or 8L:16D photoperiods, but the fish did not develop smolt morphological characteristics. In our study, it is unlikely that growth affected the changes in smoltification indices in chinook salmon because the differences in size among the fish at each sampling period were small.

Plasma Thyroxine

A long-day photoperiod has clearly been implicated as a cue for salmon to begin smoltification (Baggerman 1963; Grau et al. 1982; Higgins 1985; Duston and Saunders 1990). McCormick et al. (1987) reported results similar to our study: peak plasma thyroxine levels in Atlantic salmon were higher for fish raised under an increasing photoperiod than those raised under 24 h light. Increasing photoperiods have been used to increase thyroid function in rainbow trout (Eales 1965) and triiodothyronine and thyroxine levels in Atlantic salmon (Saunders et al. 1989). While long days are beneficial and necessary for smoltification, 24 h light or a short, unchanging photoperiod is clearly detrimental to this process. Hoffnagle and Fiv-

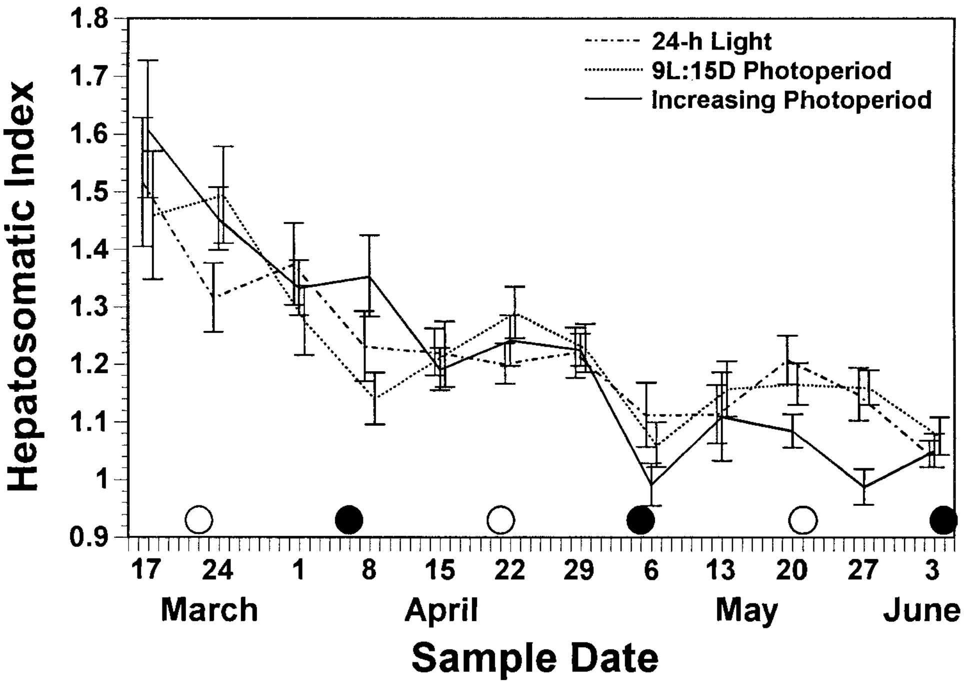

FIGURE 5.—Changes in mean (±SE) hepatosomatic index during smoltification in chinook salmon exposed to 24 h light, a constant 9L:15D photoperiod, or a naturally increasing photoperiod. New and full moon dates are indicated by filled and open circles, respectively.

izzani (1990) found that fish held indoors (constant, short photoperiod) had lower peak thyroxine levels than fish held outdoors when each group was exposed to novel water.

A ''photoperiod–endocrine axis'' has been suggested as a means of photoperiodic control of smoltification through the effects of melatonin on prolactin, thyroxine, and growth hormone (Komourdjian et al. 1976a; Thorpe 1982). Plasma prolactin levels decrease in Atlantic and coho salmon early in smoltification, and prolactin appears to have an osmoregulatory function which adapts salmon to freshwater (Prunet et al. 1985, 1989; Young et al. 1989). Young et al. (1989) showed that prolactin levels dropped dramatically in Atlantic salmon when growth hormone, thyroxine, cortisol, and gill Na^+K^+-ATPase activity all increased. Melatonin increases prolactin production, and production of melatonin in the pineal is inhibited by light. As photoperiod increases, melatonin production by the pineal gland decreases (Zachmann et al. 1992), causing prolactin levels to decrease as well. Weber and Smith (1980) found that melatonin inhibits thyroid activity by acting on the pituitary, and McKeown (1984) thought that melatonin may also depress plasma cortisol levels as well. Therefore, it appears that increasing day length causes a decrease in melatonin; this decreases prolactin levels, allowing plasma thyroxine, growth hormone, and cortisol levels to increase.

Plasma Cortisol

Plasma cortisol levels also rise during smoltification (Specker and Schreck 1982). The results of our study differ from those of Stefansson et al. (1989), who found that cortisol levels increased identically in Atlantic salmon raised under continuous light, 16L:8D, and 8L:16D photoperiods. However, McCormick et al. (1987) found that 24 h light inhibited increases in salinity tolerance and gill Na^+K^+-ATPase activity in Atlantic salmon, while those fish reared under an increasing photoperiod showed normal increases in salinity tolerance and gill Na^+K^+-ATPase activity. These data agree with those of our study because it appears that cortisol increases salinity tolerance and gill Na^+K^+-ATPase activity in salmonids (Richman and Zaugg 1987; Madsen 1990; Franklin et al. 1992). Similar to its influence on plasma thyroxine levels, there is evidence that the pineal gland and melatonin may also influence plasma cortisol lev-

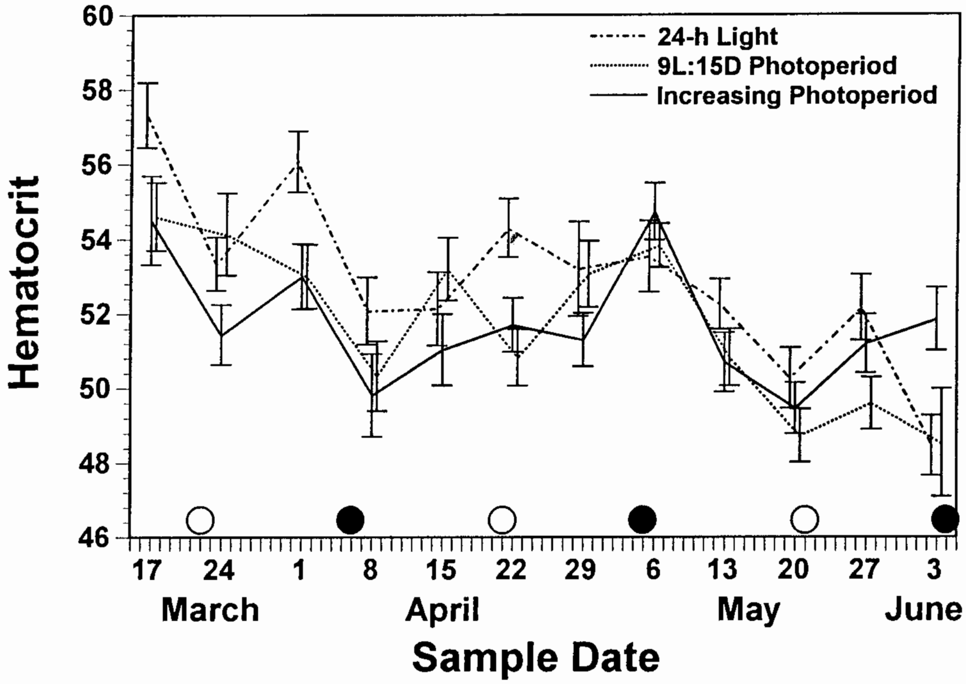

FIGURE 6.—Changes in mean (±SE) hematocrit during smoltification in chinook salmon exposed to 24 h light, a constant 9L:15D photoperiod, or a naturally increasing photoperiod. New and full moon dates are indicated by filled and open circles, respectively.

els. McKeown (1984) concluded that this may be a means for the pineal gland to indirectly affect osmoregulation in fish by influencing plasma cortisol levels.

Hepatosomatic Index

A great amount of activity occurs in the liver during smoltification. Mitochondrial activity increases (Blake et al. 1984), and shifts in liver mitochondrial and plasma protein profiles (Bradley and Rourke 1984) indicate a change in liver physiology during smoltification. Liver lipid and glycogen levels (Woo et al. 1978) and glycogen and fatty acid synthesis in the liver decrease and liver glycogen phosphorylase and lipase activities increase (Woo et al. 1978; Sheridan et al. 1985; Plisetskaya et al. 1988). Decreases in lipid and glycogen levels would probably cause decreases in liver weight, thus decreasing HSI in smolting salmon. Sheridan (1986) found that treatment of coho salmon with either thyroxine or cortisol caused a decrease in total liver lipid concentration but did not cause significant decreases in total liver mass. Soengas et al. (1992) found that treatment of rainbow trout with either cortisol or triiodothyronine and thyroxine caused a decrease in liver glycogen and glucose. Our study shows simultaneous increases in both thyroxine and cortisol in the salmon exposed to a natural photoperiod, which was also the group with the greatest decrease in HSI. It is possible that these hormones could combine to exert a sufficient synergistic effect to induce this depletion.

Hematocrit

The changes in hematocrit seen in this study further reflect the inhibitory effects of 24-h-light or unchanging photoperiods. Stefansson et al. (1989) found that hematocrit increased in Atlantic salmon during smoltification. They hypothesized that this is due to changes in respiratory capacity (Higgins 1985) caused by changes in hemoglobin (Koch 1982) and swelling of erythrocytes (Soivio and Nikinmaa 1981). Seawater has about 20% less oxygen solubility than freshwater (Wetzel 1983). To counter this, it is logical that salmon hematocrit should increase as the fish smoltify, thus increasing the oxygen carrying capacity of their blood. Benditt et al. (1941) found that when adult Atlantic salmon reentered freshwater their blood became diluted due to a decrease in the number of red blood cells which reduced the oxygen carrying ca-

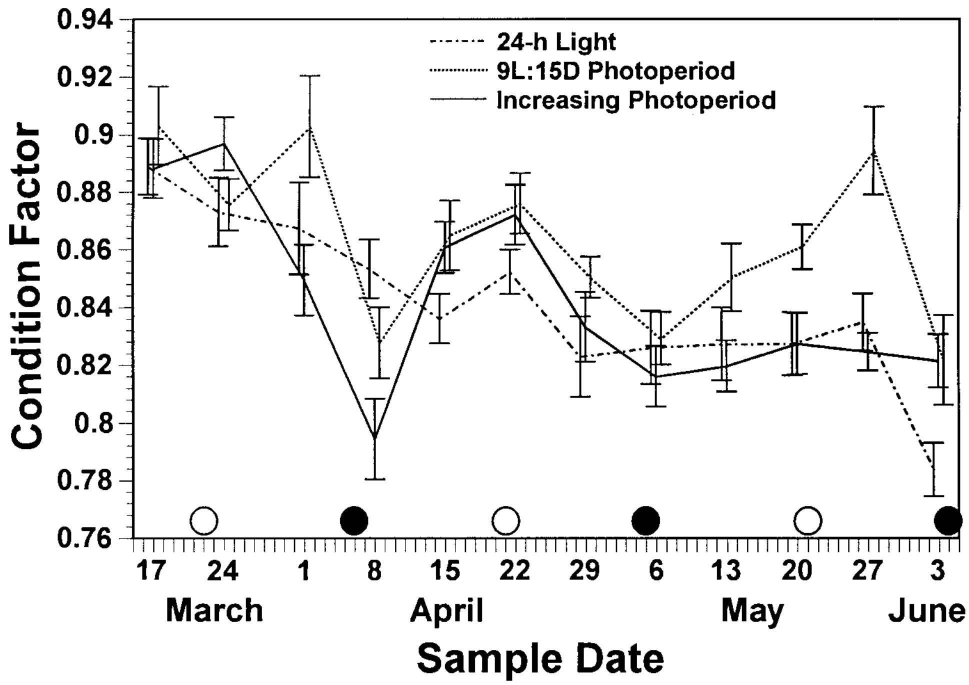

FIGURE 7.—Changes in mean (±SE) condition factor during smoltification in chinook salmon exposed to 24 h light, a constant 9L:15D photoperiod, or a naturally increasing photoperiod. New and full moon dates are indicated by filled and open circles, respectively.

pacity by 28%. Mean hematocrit levels in salmon exposed to 24 h light were significantly higher than those of the other two groups on two of the first six sampling dates and were never significantly lower throughout the study. It may be that the sudden increase in photoperiod, from 9L:15D to 24 h light, at the beginning of the experiment was sufficient to induce this early difference in hematocrit.

Condition Factor

Condition factor has long been known to decrease in smolting salmon (Hoar 1939; Wedemeyer et al. 1980). This was seen in our chinook salmon, but not to the extent reported in Atlantic (Komourdjian et al. 1976b; Virtanen and Soivio 1985; McCormick et al. 1987) and coho salmon (Barton et al. 1985). Our study supports Komourdjian et al. (1976b), who showed that long-day photoperiods induce decreases in *K*. The two groups exposed to long day lengths, 24-h and natural photoperiod, had the only decreases in *K*. Both Saunders et al. (1985) and McCormick et al. (1987) found that *K* decreased for Atlantic salmon raised under an increasing photoperiod, while *K* for fish raised under 24 h light remained nearly constant. Bjornsson et al. (1989) found that *K* increased for Atlantic salmon held under 24 h light and constant temperature. It is possible that the change to 24 h light at the beginning of our study came when the fish were large enough to be stimulated by a naturally increasing photoperiod. Duston and Saunders (1990) demonstrated that groups of Atlantic salmon that had their photoperiod increased earlier showed an earlier decrease in *K* than those fish whose photoperiod was increased later. They also showed that *K* did not decrease in fish held under constant, short days. In our study, fish in the 9L:15D photoperiod group showed no significant trend in *K* over the sampling period.

Lunar Cycles

Plasma thyroxine and cortisol levels in these chinook salmon showed some coincidental changes that culminated in the peaks seen in both hormones in the 24-h-light and natural-photoperiod groups on both 29 April and 27 May, dates of the last quarter moon, and decreases on the dates of the following new moons (6 May and 3 June). Mean plasma cortisol and thyroxine levels in the 9L:15D group also showed an increase on 29 April and a decrease on the following sampling date. This pattern was also seen on the 1 April (last quarter moon) sampling date and its succeeding sampling date (8 April; new moon) for the plasma thyroxine samples. It appears that these hormone levels may undergo a series of changes until they are all coordinated into a single, final smoltification surge. Alternatively, mean *K* showed decreases on the new moon sampling dates, particularly in the natural-photoperiod and 9L:15D groups. Previous studies have shown peak cortisol levels to occur well after peak thyroxine levels in coho salmon (Specker and Schreck 1982; Barton et al. 1985; Young et al. 1989) and in Atlantic salmon (Virtanen and Soivio 1985).

The timing of the thyroxine peaks relative to the lunar cycle in our studies conflict with studies reporting coho salmon (Grau et al. 1981), masu (=cherry) salmon *O. masou* (Yamauchi et al. 1985) and Atlantic salmon (Boeuf and Prunet 1985) undergoing thyroxine peaks on new moon dates. However, Lin et al. (1985) found thyroxine peaks in steelhead in the last quarter (1982) and full moon (1983) phases of the lunar cycle. Boeuf et al. (1989) found no relationship between plasma thyroxine levels and the new moon phase of the lunar cycle. Youngson and Simpson (1984) found no relationship between serum thyroxine levels and lunar cycle in wild or captive Atlantic salmon. Jonsson and Ruud-Hansen (1985) reported that neither streamflow, sky cover, nor lunar cycle were significantly related to the yearly variation in the timing of the Atlantic salmon smolt migration. Further confounding this, Mason (1975) found that coho salmon fry tend to migrate downstream during the new moon but that peak downstream movement of coho smolts occurred on the full moon. It is likely that the timing of the hormone peaks during smoltification are stock-specific and may be related to the distance to be traveled before reaching saltwater (i.e., stocks from coastal streams will differ from inland stocks, which must migrate hundreds of kilometers to the ocean). Boeuf et al. (1989) reported that growth hormone in Atlantic salmon from a spawning site far from the ocean ($>$1,000 km) increased earlier than in a salmon stock spawning close to the ocean. Therefore, it seems that the precise timing of these physiological, behavioral, and morphological changes is likely to vary among stocks, as well as species, of salmon.

Conclusion

Environmental conditions in hatcheries allow salmon to grow rapidly but often lack all of the environmental cues available to wild salmon for the stimulation of smoltification, such as changing photoperiod, water temperature, and chemistry. Thorpe (1991) concluded that development of juvenile salmon is genetically determined but is influenced by the environment. The results of our study further support the hypothesis of Nishioka et al. (1985) that hatchery rearing of young salmonids may not provide a sufficiently changing environment for stimulation of the thyroid gland and thus proper induction of smoltification. Most previous studies were conducted at hatcheries where salmon were raised outdoors, which exposed them to natural photoperiods. The present study and that of Hoffnagle and Fivizzani (1990) demonstrate that changing photoperiod and water chemistry can have additive effects on smoltification indices and are necessary for a complete, coordinated smoltification process. This lack of environmental change in hatcheries may be the basis for the lack of smoltification-associated increases in plasma thyroxine in nonexperimental (control) chinook salmon in previous studies. We have observed rapid fivefold (Hoffnagle 1994) and sevenfold (Hoffnagle and Fivizzani 1990) increases in plasma thyroxine levels in chinook salmon raised in well water and under a constant photoperiod and then transferred to an outdoor site with a novel water chemistry. This may be the result of previously understimulated fish being suddenly exposed to environmental cues which trigger smoltification. Photoperiodic changes can be used as a management tool to induce or delay smoltification. Hatchery managers should account for photoperiodic effects when raising salmon and avoid the use of haphazard photocycles or continuous light. Stefansson et al. (1990) reported that a dual photoperiod, consisting of low-intensity background illumination combined with a natural photoperiod, improved growth rate without seriously affecting parr–smolt transformation in Atlantic salmon, as determined by seawater growth. This dual photoperiod should be examined more fully to determine its effects on behavior and other smoltification indices.

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References

Baggerman, B. 1963. The effect of TSH and antithyroid substances on salinity preference and thyroid activity in juvenile Pacific salmon. Canadian Journal of Zoology 41:307–319.

Barton, B. A., C. B. Schreck, R. D. Ewing, A. R. Hemmingsen, and R. Patiño. 1985. Changes in plasma cortisol during stress and smoltification in coho salmon, *Oncorhynchus kisutch.* General and Comparative Endocrinology 59:468–471.

Benditt, E., P. Morrison, and L. Irving. 1941. The blood of the Atlantic salmon during migration. Biological Bulletin 80:429–440.

Bjornsson, B. T., H. Thorarensen, T. Hirano, T. Ogasawara, and J. B. Kristinsson. 1989. Photoperiod and temperature affect plasma growth hormone levels, growth, condition factor and hypoosmoregulatory ability of juvenile Atlantic salmon (*Salmo salar*) during parr–smolt transformation. Aquaculture 82: 77–91.

Blake, R. L., F. L. Roberts, and R. L. Saunders. 1984. Parr–smolt transformation of Atlantic salmon (*Salmo salar*): activities of two respiratory enzymes and concentrations of mitochondria in the liver. Canadian Journal of Fisheries and Aquatic Sciences 41: 199–203.

Boeuf, G., P. Y. Le Bail, and P. Prunet. 1989. Growth hormone and thyroid hormones during Atlantic salmon, *Salmo salar* L., smolting and after transfer to seawater. Aquaculture 82:257–268.

Boeuf, G., and P. Prunet. 1985. Measurements of gill (Na^+-K^+)-ATP-ase activity and plasma thyroid hormones during smoltification in Atlantic salmon (*Salmo salar* L.). Aquaculture 45:111–119.

Bradley, T. M., and A. W. Rourke. 1984. An electrophoretic analysis of plasma proteins from juvenile *Oncorhynchus tshawytscha* (Walbaum). Journal of Fish Biology 24:703–709.

Clarke, W. C., H. Lundqvist, and L. O. Eriksson. 1985. Accelerated photoperiod advances seasonal cycle of seawater adaptation in juvenile Baltic salmon, *Salmo salar* L. Journal of Fish Biology 26:29–35.

Clarke, W. C., J. E. Shelbourne, and J. R. Brett. 1978. Growth and adaptation to sea water in 'underyearling' sockeye (*Oncorhynchus nerka*) and coho (*O. kisutch*) salmon subjected to regimes of constant or changing temperature and day length. Canadian Journal of Zoology 56:2413–2421.

Duston, J., and J. D. E. Knox. 1992. Acclimation of Atlantic salmon (*Salmo salar*) parr to seawater in autumn: stimulatory effect of a long photoperiod. Aquaculture 103:341–358.

Duston, J., and R. L. Saunders. 1990. The entrainment role of photoperiod on hypoosmoregulatory and growth-related aspects of smolting in Atlantic salmon (*Salmo salar*). Canadian Journal of Zoology 68: 707–715.

Eales, J. G. 1965. Factors influencing seasonal changes in thyroid activity in juvenile steelhead trout, *Salmo gairdneri.* Canadian Journal of Zoology 43:719–729.

Folmar, L. C., and W. W. Dickhoff. 1981. Evaluation of some physiological parameters as predictive indices of smoltification. Aquaculture 23:309–324.

Franklin, C. E., W. Davison, and M. E. Forster. 1992. Seawater adaptability of New Zealand's sockeye (*Oncorhynchus nerka*) and chinook salmon (*O. tshawytscha*): physiological correlates of smoltification and seawater survival. Aquaculture 102:127–142.

Grau, E. G., W. W. Dickhoff, R. S. Nishioka, H. A. Bern, and L. C. Folmar. 1981. Lunar phasing of the thyroxine surge preparatory to seaward migration of salmonid fish. Science 211:607–609.

Grau, E. G., J. L. Specker, R. S. Nishioka, and H. A. Bern. 1982. Factors determining the occurrence of the surge in thyroid activity in salmon during smoltification. Aquaculture 28:49–57.

Higgins, P. J. 1985. Metabolic differences between Atlantic salmon (*Salmo salar*) parr and smolts. Aquaculture 45:33–53.

Hoar, W. S. 1939. The weight–length relationship of the Atlantic salmon. Journal of the Fisheries Research Board of Canada 4:441–460.

Hoffnagle, T. L. 1994. Smoltification and imprinting in introduced, land-locked chinook salmon (*Oncorhynchus tshawytscha*). Doctoral dissertation. University of North Dakota, Grand Forks.

Hoffnagle, T. L., and A. J. Fivizzani. 1990. Stimulation of plasma thyroxine levels by novel water chemistry during smoltification in chinook salmon (*Oncorhynchus tshawytscha*). Canadian Journal of Fisheries and Aquatic Sciences 47:1513–1517.

Jonsson, B., and J. Ruud-Hansen. 1985. Water temperature as the primary influence on timing of seaward migrations of Atlantic salmon (*Salmo salar*) smolts. Canadian Journal of Fisheries and Aquatic Sciences 42:593–595.

Kleinbaum, D. G., and L. L. Kupper. 1978. Applied regression analysis and other multivariable methods. Doxbury Press, Boston.

Koch, H. J. A. 1982. Hemoglobin changes with size in the Atlantic salmon (*Salmo salar* L.). Aquaculture 28:231–240.

Komourdjian, M. P., R. L. Saunders, and J. C. Fenwick. 1976a. Evidence for the role of growth hormone as a part of a 'light–pituitary axis' in growth and smoltification of Atlantic salmon (*Salmo salar*). Canadian Journal of Zoology 54:544–551.

Komourdjian, M. P., R. L. Saunders, and J. C. Fenwick. 1976b. The effect of porcine somatotropin on growth and survival in seawater of Atlantic salmon (*Salmo salar*) parr. Canadian Journal of Zoology 54: 531–535.

Lin, R. J., R. J. Rivas, R. S. Nishioka, E. G. Grau, and H. A. Bern. 1985. Effects of feeding triiodothyronine (T_3) on thyroxine (T_4) levels in the steelhead trout. Aquaculture 45:133–142.

Lundqvist, H. 1980. Influence of photoperiod on growth in Baltic salmon (*Salmo salar* L.) with special reference to the effect of precocial sexual maturation. Canadian Journal of Zoology 58:940–944.

Madsen, S. S. 1990. Enhanced hypoosmoregulatory response to growth hormone after cortisol treatment in immature rainbow trout, *Salmo gairdneri.* Fish Physiology and Biochemistry 8:271–279.

Mason, J. C. 1975. Seaward movement of juvenile fishes, including lunar periodicity in the movement of coho salmon (*Oncorhynchus kisutch*) fry. Journal of the Fisheries Research Board of Canada 32:2542–2547.

McCormick, S. D., R. L. Saunders, E. B. Henderson, and P. R. Harmon. 1987. Photoperiod control of parr–smolt transformation in Atlantic salmon (*Salmo salar*): changes in salinity tolerance, gill Na-K-ATPase activity and plasma thyroid hormones. Canadian Journal of Fisheries and Aquatic Sciences 44:1462–1468.

McKeown, B. A. 1984. Fish migration. Timber Press, Portland, Oregon.

Nishioka, R. S., G. Young, H. A. Bern, W. Jochimsen, and C. Hiser. 1985. Attempts to intensify the thyroxin surge in coho and king salmon by chemical stimulation. Aquaculture 45:215–225.

Plisetskaya, E. M., P. Swanson, M. G. Bernard, and W. W. Dickhoff. 1988. Insulin and coho salmon (*Oncorhynchus kisutch*) during the parr to smolt transformation. Aquaculture 72:151–164.

Prunet, P., G. Boeuf, J. P. Bolton, and G. Young. 1989. Smoltification and seawater adaptation in Atlantic salmon (*Salmo salar*): plasma prolactin, growth hormone and thyroid hormones. General and Comparative Endocrinology 74:355–364.

Prunet, P., G. Boeuf, and L. M. Houdebine. 1985. Plasma prolactin levels in rainbow trout during the adaptation to different salinities. Journal of Experimental Zoology 235:187–196.

Richman, N. H., III, and W. S. Zaugg. 1987. Effects of cortisol and growth hormone on osmoregulation in pre- and desmoltified coho salmon (*Oncorhynchus kisutch*). General and Comparative Endocrinology 65:189–198.

Rottiers, D. V. 1992. Effects of day length and cleaning regimen on the growth of yearling parr Atlantic salmon. Progressive Fish-Culturist 54:69–72.

Saunders, R. L., and E. B. Henderson. 1970. Influence of photoperiod on smolt development and growth of Atlantic salmon (*Salmo salar*). Journal of the Fisheries Research Board of Canada 27:1295–1311.

Saunders, R. L., and E. B. Henderson. 1988. Effects of constant day length on sexual maturation and growth of Atlantic salmon (*Salmo salar*) parr. Canadian Journal of Fisheries and Aquatic Sciences 45:60–64.

Saunders, R. L., E. B. Henderson, and P. R. Harmon. 1985. Effects of photoperiod on juvenile growth and smolting of Atlantic salmon and subsequent survival and growth in sea cages. Aquaculture 45:55–66.

Saunders, R. L., J. L. Specker, and M. P. Komourdjian. 1989. Effects of photoperiod on growth and smolting in juvenile Atlantic salmon (*Salmo salar*). Aquaculture 82:103–117.

Sheridan, M. A. 1986. Effects of thyroxin, cortisol, growth hormone and prolactin on lipid metabolism of coho salmon, *Oncorhynchus kisutch,* during smoltification. General and Comparative Endocrinology 64:220–238.

Sheridan, M. A., N. Y. S. Woo, and H. A. Bern. 1985. Changes in the rates of glycogenesis, glycogenolysis, lipogenesis and lipolysis in selected tissues of the coho salmon (*Oncorhynchus kisutch*) associated with parr–smolt transformation. Journal of Experimental Zoology 236:35–44.

Soengas, J. L., P. Rey, G. Rozas, M. D. Andres, and M. Aldegunde. 1992. Effects of cortisol and thyroid hormone treatment on the glycogen metabolism of selected tissues of domesticated rainbow trout, *Oncorhynchus mykiss.* Aquaculture 101:317–328.

Soivio, A., and M. Nikinmaa. 1981. The swelling of erythrocytes in relation to the oxygen affinity of the blood of the rainbow trout, *Salmo gairdneri* Richardson. Pages 101–119 *in* A. D. Pickering, editor. Stress and fish. Academic Press, London.

Specker, J. L., and C. B. Schreck. 1982. Changes in plasma corticosteroids during smoltification of coho salmon, *Oncorhynchus kisutch.* General and Comparative Endocrinology 46:53–58.

Stefansson, S. O., T. Hansen, and G. L. Taranger. 1990. The influences of light on growth and smolting of Atlantic salmon (*Salmo salar*): effects of spectral composition, intensity and photoperiod. Canadian Technical Report of Fisheries and Aquatic Sciences 1761:79–84.

Stefansson, S. O., G. Naevdal, and T. Hansen. 1989. The influence of three unchanging photoperiods on growth and parr–smolt transformation in Atlantic salmon, *Salmo salar* L. Journal of Fish Biology 35: 237–247.

Thorpe, J. E. 1982. Migration in salmonids, with special reference to juvenile movements in freshwater. Pages 86–97 *in* E. L. Brannon and E. O. Salo, editors. Proceedings of the salmon and trout migratory behavior symposium. University of Washington, School of Fisheries, Seattle.

Thorpe, J. E. 1991. Acceleration and deceleration effects of hatchery rearing on salmonid development, and their consequences for wild stocks. Aquaculture 98:111–118.

Virtanen, E., and A. Soivio. 1985. The patterns of T_3, T_4, cortisol and Na+-K+-ATP-ase during smoltification of hatchery-reared *Salmo salar* and comparison with wild smolts. Aquaculture 45:97–109.

Wagner, H. H. 1974. Photoperiod and temperature regulation of smolting in steelhead trout (*Salmo gairdneri*). Canadian Journal of Zoology 52:219–234.

Weber, L. J., and J. R. Smith. 1980. Possible role of the pineal gland in migratory behavior of salmonids. Pages 313–320 *in* W. J. McNeil and D. C. Himsworth, editors. Salmonid ecosystems of the north Pacific. Oregon State University Press, Corvallis.

Wedemeyer, G. A., R. L. Saunders, and W. C. Clarke. 1980. Environmental factors affecting smoltification and early marine survival of anadromous salmonids. U.S. National Marine Fisheries Service Marine Fisheries Review 6:1–14.

Wetzel, R. G. 1983. Limnology, second edition. Saunders, Philadelphia.

Woo, N. Y. S., H. A. Bern, and R. S. Nishioka. 1978. Changes in body composition associated with smoltification and premature transfer to seawater in coho salmon (*Oncorhynchus kisutch*) and king salmon (*O. tshawytscha*). Journal of Fish Biology 13:421–428.

Yamauchi, K., and five coauthors. 1985. Physiological and behavioral changes occurring during smoltification in the masu salmon, *Oncorhynchus masou.* Aquaculture 45:227–235.

Young, G., B. T. Bjornsson, P. Prunet, R. J. Lin, and H. A. Bern. 1989. Smoltification and seawater adaptation in coho salmon (*Oncorhynchus kisutch*): plasma prolactin, growth hormone, thyroid hormones, and cortisol. General and Comparative Endocrinology 74:335–345.

Youngson, A. F., and T. H. Simpson. 1984. Changes in serum thyroxine levels during smolting in captive and wild Atlantic salmon, *Salmo salar* L. Journal of Fish Biology 24:29–39.

Zachmann, A., S. C. M. Knijff, M. A. Ali, and M. Anctil. 1992. Effects of photoperiod and different intensities of light exposure on melatonin levels in the blood, pineal organ and retina of brook trout (*Salvelinus fontinalis* Mitchill). Canadian Journal of Zoology 70:25–29.

Zaugg, W. S., and H. H. Wagner. 1973. Gill ATPase activity related to parr–smolt transformation and migration in steelhead trout (*Salmo gairdneri*): influence of photoperiod and temperature. Comparative Biochemistry and Physiology 45B:955–965.

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CDFW-IFP-001

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Standard Operating Procedure for Critical Riffle Analysis

for Fish Passage in California

CDFW-IFP-001

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Acknowledgements

This standard operating procedure (SOP) represents the protocol for critical riffle analysis (CRA) of the California Department of Fish and Wildlife (Department; previously known as California Department of Fish and Game) Water Branch Instream Flow Program (IFP). It is intended to replace the original 2012 document, which received updates in 2013 and 2015. The process in this SOP draws from current methods used by the Oregon Department of Fish and Wildlife (ODFW). Modifications are made to ODFW methods in both procedural scope and in the application of regional information relevant to California. The overall concept of the procedure in this SOP is based on information in *Determining Stream Flows for Fish Life* presented by Ken Thompson at the *Instream Flow Requirements Workshop* on March 15-16, 1972 (Thompson 1972). This SOP was developed by the Marine Pollution Studies Laboratory Quality Assurance Team. The Department IFP provided Microsoft Excel® (Excel) spreadsheet technical assistance. Technical review of this document was provided by IFP staff, and members of the ODFW Water Quality and Quantity Program. This document has been reviewed by the Department Office of the General Counsel.

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Abbreviations and Acronyms

CRA	Critical Riffle Analysis
Department	California Department of Fish and Wildlife
IFP	Instream Flow Program
ODFW	Oregon Department of Fish and Wildlife
SOP	Standard Operating Procedure

Introduction

This document serves as the SOP for CRA studies for the Department's IFP. It may be used in conjunction with other IFP SOPs. Instructions are provided for:

- Project planning considerations:
  - Project timing
  - Site selection
  - Fish passage criteria
- Field methods:
  - Transect preparation
  - Data collection
- Data analysis:
  - Tabular data entry
  - Consideration of fish passage criteria
  - Graphing of results
  - Comparison of results against fish passage criteria

Scope of Application

This SOP provides procedural reference for Department staff conducting CRA, when site conditions and research objectives indicate that CRA is an appropriate methodology. It is also intended as an informational resource for other state and federal agencies, nongovernmental organizations, private contractors, and other organizations throughout California. Fish passage criteria cited in this document are specific to California and should not be extrapolated beyond the state borders.

This SOP applies only to wadeable streams having low gradient riffles with less than 4% gradient and substrates dominated by gravel and cobble. This procedure is used to identify flows that support physical movement of salmonids through critical riffle sites. Other factors that may be important to evaluate overall migratory success include length of riffle, availability of rest areas, condition of fish, water temperature, and others.

This SOP is not applicable to high gradient riffles with greater than 4% gradient and boulder dominated substrates (Flosi et al. 1998). It does not apply to river or stream channels that do not have riffles, such as those dominated by silt and sand substrates with particle sizes less than 0.1 inches. Finally, this procedure is not applicable to culverts, weirs, bedrock ledges, or anticlines with associated drops.

Note: Safety should always be a primary concern when conducting CRA. Do not conduct sampling when field conditions are unsafe.

What is Critical Riffle Analysis?

The CRA methodology is used to identify stream flow rates necessary for the passage of salmon and trout through critical riffles. Riffles are habitat units in streams and rivers with relatively shallow depth and swiftly flowing turbulent water. They serve multiple functions in the ecological processes of cold water streams and rivers, and are an integral link in the life histories of salmon and trout. Many species of aquatic macroinvertebrates develop and grow in riffles, which provide a food source for salmonids. Riffles also provide salmonids with well-mixed oxygenated water and escape shelter from predators.

Critical riffles are particularly shallow and sensitive to changes in stream flow. Changes in stream flow and associated water depth may limit the hydrologic connectivity of river habitats and impede critical life history tactics of salmon and trout. In such cases, the critical riffle may become a potential barrier to upstream and downstream salmonid passage. Critical riffles may prevent adults from moving to and from spawning areas, prevent smolts from migrating downstream to staging areas in brackish waters of lagoons and estuaries before entering the ocean, and prevent rearing juvenile salmonids (e.g., steelhead) from being able to move between adequate summer freshwater rearing habitats.

Method Overview

Stream flow rates for salmon and trout passage are determined by locating a critical riffle, identifying a transect along the riffle’s shallowest course from bank to bank, measuring water depth at a set interval across the transect, and visiting the site over a range of flow. Adequate water depths over a sufficient width of the transect are necessary to identify flows for passage of adult and juvenile salmonids through critical riffle sites. Field measurements are compared to target species and lifestage water depth criteria (see *Section 3* for more information). After a minimum of three to six field sampling events have been completed along the transect over a wide range of flows, a graph of stream flow rates versus the corresponding percent of transect meeting the minimum depth criteria for the species and lifestages can be used to determine flow rates necessary for passage.

Section 1: Considerations for Project Planning

Before collecting data for CRA, it is important to identify both the appropriate timing of the sampling events and the appropriate site. The timing of the sampling events should be linked to the target species and lifestage.

CRA is conducted by establishing a transect across the critical riffle and collecting depth measurements along that transect. This SOP uses depth criteria for a target species and lifestage to determine flow rates necessary for passage of that species through critical riffles. The project manager may decide to collect stream velocity measurements along the critical riffle transect and assess results in relation to a target species' maximum velocity tolerances. Please consult the Department IFP for more information about these procedures before planning such a project.

Crew safety is of paramount importance; ensure that the river can be safely sampled during the highest flow point. Contact the Department IFP for project planning assistance, as needed.

1.1 Project Timing

Data for CRA are collected during three to six field sampling events, typically during the receding limb of the hydrograph. The sampling events should be timed to capture the full range of discharges for the passage of the target species and lifestages. Ideally, the first sampling event would be at the highest wadeable flow, with subsequent sampling events taken as flow decreases.

Development of a flow exceedance probability chart based upon unimpaired flow conditions for the period of record may be useful to identify flows for field sampling. The exceedance probability chart will indicate the percentage of time that a stream flow rate is likely to equal or exceed a value of interest. A good project planning starting point is to identify the 20%, 50%, and 80% exceedance flows for field sampling. To develop a robust relationship between salmon and trout passable criteria and flows, additional sampling events are needed. These additional sampling events for data collection will most likely be at flows between the 20% to 80%

exceedance probability range, depending on site-specific conditions such as channel morphology, substrate, and flow relationships.

When identifying flows for field sampling, Department staff should consider that each field event must have at least one depth measurement along the CRA transect that meets or exceeds the target species and lifestage depth criteria (see Section 3.2) in order for that sampling event to be used in the analysis.

It may be useful to create a lifestage periodicity table indicating when target species migration occurs in the river of interest. This table may be used to determine what exceedance flows likely occur during migration as well as help to plan the timing of field work so that sampling events encapsulate the appropriate range of flows for the species and lifestage of interest. Contact the Department IFP for assistance developing a flow exceedance probability chart or lifestage periodicity table, as needed. The Department SOP CDFW-IFP-005 (CDFW 2013a) on how to develop flow exceedance probability charts is available on the Department's IFP web site.

1.2 Site Selection: Locating the Critical Riffle

The identification of sampling sites for CRA should be made collaboratively by Department staff familiar with the study area. In some cases, it may be difficult to identify or agree collaboratively on the most critical riffle (i.e., most water depth-sensitive riffle) in a stream reach by visual observation alone. In these instances, it may be necessary to collect depth measurements at multiple critical riffle sites, including the maximum thalweg depth along the shallowest course of each riffle, in order to identify the most depth-sensitive critical riffles and their respective flows for salmonid passage.

Section 2: Field Procedures

Once a critical riffle has been identified for CRA, the transect is established, marked, and photographed. During the initial CRA sampling event, the critical riffle transect is located and marked with head- and tailpins and flagged with site information. CRA also requires a discharge measurement, which may be obtained either from an appropriate stream gage, or by measuring discharge as outlined in the Department discharge SOP CDFW-IFP-002 (CDFW 2013b).

The field data sheet can be found online at the Department IFP documents page: https://www.wildlife.ca.gov/Conservation/Watersheds/Instream-Flow/SOP.

2.1 Equipment List

Field crews should pack the following equipment for sampling:

- Stadia rod (engineering grade rod capable of measuring 1/10 ft and 1/100 ft)
- Fiberglass measuring tape (100-300 ft)
- Staff gage
- 0.5 in x 2.5 - 4 ft rebar (2-10 pieces depending on site)
- Hammer or mallet
- GPS unit
- Field data sheets (available at https://www.wildlife.ca.gov/Conservation/Watersheds/Instream-Flow/SOP)
- Pencils
- Flagging and permanent marker
- Camera
- Calculator
- Small carabiners or spring clamps (5-10; optional)

Note: If discharge will be measured in the field, crews should also pack equipment as listed in the Department discharge SOP CDFW-IFP-002.

2.2 Establishing the Transect

Establish a transect running along the shallowest course of the riffle from bank to bank using rebar and a measuring tape. This transect will rarely be linear, but should instead follow the contours of the riffle along its shallowest course from bank to bank (*Figures 1-3*). The critical riffle transect is established during the first sampling event, and is then used repeatedly for each subsequent sampling event. Once the transect is identified, markers are placed at the wetted edge on each bank marking the headpin and tailpin, and along the course of the shallowest contour. During subsequent sampling events, the course of the shallowest contour across the riffle should be re-identified and verified with depth measurements to confirm transect location.

<u>Step 1</u>: Set the headpin for the transect on the left bank of the river looking upstream. The headpin serves as the starting point for each critical riffle measurement, starting from zero feet.

<u>Step 2:</u> Attach a flag to the headpin. This flag is marked with project and site identification information.

<u>Step 3:</u> Set the tailpin adjacent to the edge of the critical riffle on the right bank of the river when facing upstream.

<u>Step 4:</u> Attach a wind-up, light weight measuring tape to the base of the headpin. The tape should be of sufficient length for the site (e.g., 100-ft, 200-ft, 300-ft). The tape should display 1/10-ft measurements.

<u>Step 5:</u> Work across the riffle, following the contour of shallowest course (*Figure 2* and *3*). Use a stadia rod to locate the shallowest depths. Hammer in rebar while working across the riffle. Secure the measuring tape along the riffle contour by wrapping tape around rebar, or by using small carabiners or spring clamps to hold the tape in place.

<u>Step 6:</u> Continue to work across the riffle until the tailpin is reached. Attach the other end of the measuring tape to the base of the tailpin.

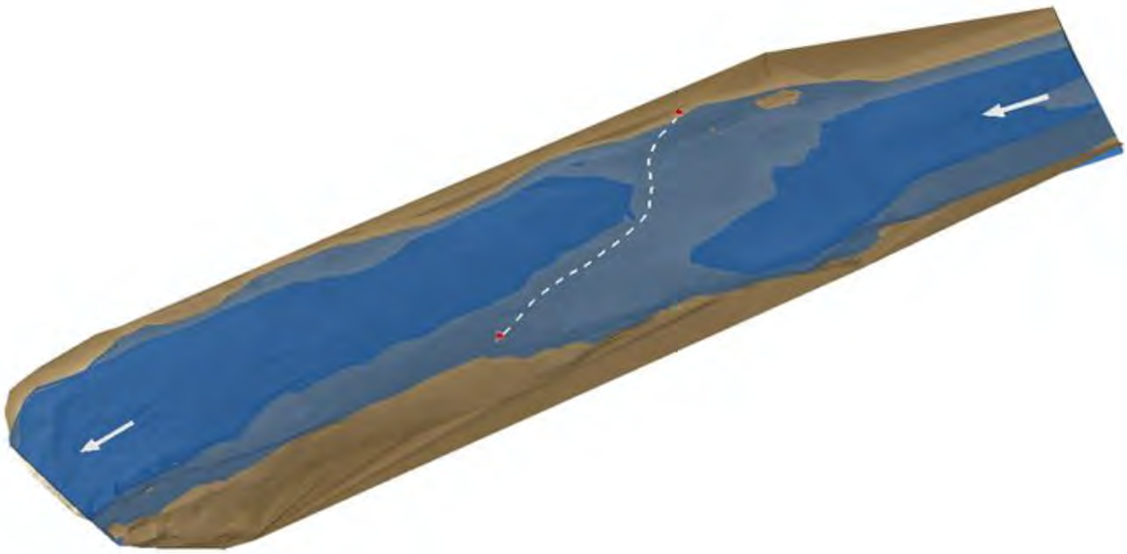

Figure 1. Example of passage transect delineation across a critical riffle

Figure 2. Example of a critical riffle transect that follows the shallowest course from bank to bank

Figure 3. Additional example of a critical riffle transect following the shallowest course from bank to bank

2.3 Data Collection

Once the CRA transect has been set up, data are collected.

Step 1: If not already in place, set up the measuring tape along the contour of the riffle's shallowest course, as described in *Section 2.2*.

Step 2: During each sampling event, document the transect with multiple photographs, taken while facing upstream and downstream.

Step 3: Populate the CRA field data sheet with the following:

- Stream name (e.g., Candice River)
- Reach (e.g., Lower Reach)
- Riffle name (e.g., CR5)
- Site description (e.g., Wide transverse riffle near pump #34)
- GPS coordinates or waypoint (e.g., N38° 53.331, W121° 17.092; or wpt. 22)
- Total length of the transect from headpin to tailpin (HP to TP; i.e., total transect length following the shallowest course from bank to bank)

- Left bank wetted edge (LBWE; i.e., the distance on the tape measure where the wetted edge exists on the left bank)
- Right bank wetted edge (RBWE; i.e., the distance on the tape measure where the wetted edge exists on the right bank)
- Sampling date
- Evaluator (i.e., initials of staff conducting CRA)
- Recorder (i.e., initials of staff recording data)
- Photo file range
- Time start (i.e., the time when sampling starts)
- Staff gage start (i.e., the stage height from a nearby staff gage when sampling starts). Staff gage may be a temporary gage that is installed just for purposes of doing the CRA, or it may be a nearby permanent gage if one exists. The staff gage stage height is used to assess whether fluctuations in flow occurred during the sampling event.

<u>Step 4:</u> Determine the interval size at which to measure depth along the transect by considering the total transect length. Depth should be measured a <u>minimum</u> of 20 times at regularly spaced intervals along the transect. The increment of the interval should be small enough to accurately represent changes in bed profile elevation. A minimum sampling interval of <u>one foot</u> is recommended for CRA sites with critical riffles of greater than 20 ft from bank to bank. The sampling interval for CRA sites that are less than 20 ft from bank to bank should be adjusted so as to meet the 20 minimum depth measurements. A more robust depth to flow relationship is achieved with approximately 50 depth measurements on the critical riffle (Tim Hardin, ODFW, personal communication, July 2, 2012).

<u>Step 5:</u> Using a stadia rod, measure water depth to the nearest 0.01 ft along the transect at the interval distance determined in Step 4 (*Figures 4* and *5*). Record station distance and depth in feet on the CRA field data sheet. There must be at least one depth measurement that meets or exceeds the target species and lifestage depth criteria (see *Section 3.2*) in order for the sampling event to be used in the analysis.

Note: Careful attention should be taken to record water depths at individual locations as the fish would encounter and use them. For example, the stadia rod should not be placed between two rocks to measure depth unless it appears that a fish could swim between them freely. Using the *same example, if the stadia rod fits between two rocks (without being on top of the rocks), but the upstream passage is blocked by other rocks immediately upstream or downstream within the measurement cell of the longitudinal profile of the critical riffle, then the measurement should be taken on top of, instead of between the rocks. In these cases the shallowest points (on top) of the wetted substrate should be selected if the fish could not use the depths between rocks due to passage obstructions either immediately upstream or downstream of the transect. Add any such notes to the comments section on the CRA field data sheet.*

<u>Step 6:</u> Populate closing data fields on the CRA field data sheet (i.e., end time and staff gage end depth).

<u>Step 7:</u> Ensure the staff gage remained constant during the sampling event, then remove the measuring tape from headpin and tailpin and clean up other equipment as necessary.

<u>Step 8:</u> Obtain and record discharge data on the CRA field data sheet either from an appropriate stream gage, or by conducting a site-specific discharge measurement in accordance with *CDFW-IFP-002*.

<u>Step 9:</u> Ensure that all data fields have been populated on the CRA field data sheet before leaving the site.

Figure 4. Example using stadia rod to measure depth along a critical riffle transect

Figure 5. Detail of stadia rod used to measure depth along a critical riffle transect

Section 3: Data Entry and Analysis

After each sampling event, data are entered and stored in Excel for data analysis. Depth data are compared with the target species and lifestage depth criteria for fish passage. This section discusses data entry and the criteria for fish passage, details how to determine stream flow rates for salmonid passage, and provides important considerations when applying the method.

3.1 Data Entry

After each sampling event, transfer data from the field data sheet to the corresponding spreadsheet in the Excel *Workbook for Critical Riffle Analysis for Fish Passage in California* (available online at the Department IFP SOP and QA/QC documents page: https://www.wildlife.ca.gov/Conservation/Watersheds/Instream-Flow/SOP).

Disclaimer: Due to variability in field data (e.g., station distances, transect lengths), adjustments in data entry fields may be needed to correctly populate the spreadsheet. Please contact the Department IFP for assistance. The Department is not responsible for inappropriate application of the Excel workbook or spreadsheets.

The Excel workbook contains spreadsheets to calculate the number of cells that meet the minimum depth criteria for fish passage (as outlined in *Section 3.2*) for each sampling event. See below for specific data entry instructions for transferring field data into the Excel workbook.

There are six tabs named for each of six sampling events (e.g., “CRA Passage Form 1” for sampling event one) in the online Excel workbook. The Excel workbook also contains:

1. The “Read Me” tab, which provides condensed instructions on data entry and analysis
2. The “Example Passage Form”, which is a filled-out example data form
3. The “CRA Cumulative Calculations” tab, which includes cumulative calculation data tables for the percent contiguous passable width and percent total passable width data; this is used to summarize the results from each sampling event for subsequent data analysis
4. The “Minimum Depth Criteria” tab, which includes the salmon and trout species or lifestage codes and their minimum depth criteria for passage

Using a “CRA Passage Form” tab, fill out all relevant information for the sampling event (e.g., stream name, site description, date). Specify the target species and lifestage for the CRA and enter the respective criteria into the form field. The target species and lifestage criteria (see *Section 3.2*) are selected from the “Minimum Depth Criteria” tab in the Excel workbook. For example, enter “St” in the Species/Lifestage Code and “0.7” ft depth in the Target Species Depth Criteria sections of the spreadsheet if interested in assessing flows for adult steelhead passage.

Continue entering field data into the “Distance” and “Depth” categories. The “Distance” entry is typically the position of the tape measure where the depth data were recorded. The “Width” column will be populated by the spreadsheet based upon the distance or locations of depth measurements taken. The spreadsheet will calculate the percent contiguous passable flows and percent total passable flows for each sampling event. In order for the “Percent Contiguous” calculation to be correctly summarized, the user will have to “Sum” the longest range of contiguous cells meeting the depth criteria and enter this value in the contiguous width cell on the bottom of the spreadsheet (highlighted in red).

It is important to note that the maximum transect length (i.e., maximum wetted width) must be entered in each of the (three to six) CRA Passage Forms for the percent contiguous and percent total calculations to be accurate. The maximum transect length is generally associated with the highest flow following the shallowest course from bank to bank. Maximum transect lengths should typically reach but not exceed beyond the toe of bank – the point where the streambed and bank join. The streambed is defined as that part of the channel usually not occupied by perennial terrestrial plants, but including gravel bars, and lying between the toe of each bank.

After all of the sampling events have been conducted, and data entered for all events in their individual CRA Passage Form tabs, select the “CRA Cumulative Calculations” tab. Manually enter the flows of each sampling event along with the percent of maximum transect length meeting passage depth criteria for the target species or lifestage as outlined above.

3.2 Criteria for Fish Passage

Stream width and depth criteria are used to derive flows for salmonid passage. The Department IFP has adopted two width criteria for development of flows for salmon and trout passage from Thompson (1972):

1. At least 10% of the maximum wetted transect length must be contiguous for the minimum depth criterion established for the target fish; *and*
2. A total of at least 25% of the maximum wetted transect length must be at least the minimum depth criterion established for the target fish.

The minimum water depth criteria for salmonids are outlined in *Table 1*. These criteria are based upon a literature review conducted by R2 Resources (2008) and are intended to provide protective conditions for passage. Ideally, there should be sufficient clearance underneath a fish so that contact with the streambed and abrasion are minimized, which R2 Resources (2008) considered to be 0.1 ft. When selecting the appropriate depth criteria, use the minimum depth for the adult fish if both adult and juvenile fish are known to be in the system at the same time. The Department may update the minimum depths in *Table 1* as new information is developed.

Table 1: Minimum depth criteria for adult and juvenile salmonid passage to be used in critical riffle analysis

Species	Minimum Depth (ft)
Chinook Salmon (adult)	0.9
Steelhead (adult)	0.7
Coho Salmon (adult)	0.7
Trout (adult, including 1-2+ juvenile steelhead)	0.4
Salmonid (young of year juvenile)	0.3

3.3 Graphing Results and Data Analysis

After data have been entered in the “CRA Cumulative Calculations” tab of the Excel workbook, stream flow rates for salmonid passage are determined by graphing and examining the relationship between flow and the stream width and depth criteria.

For each target species, generate two graphs:

- Flow versus percent contiguous passable width
- Flow versus percent total passable width

To determine the flows for target fish passage through the critical riffle, generate a best-fit regression line on each graph. First, find the point on the Y-axis that meets each criterion (e.g., 10% contiguous passable width), and then find where this point hits the line of best fit and its corresponding point on the X-axis. This point on the X-axis is the discharge for fish passage through this critical riffle.

Note: If there is more than one target species (as listed in Table 1) involved in the study, generate one set of graphs per target species.

3.4 Example of Data Analysis Results

A critical riffle measuring 100 ft (maximum transect length) from bank to bank along its shallowest course is being analyzed for passage of adult steelhead (minimum depth = 0.7 ft). To meet Department IFP criteria for adult steelhead passage, this riffle would need to have:

- A *contiguous* portion of at least 10 ft in length measuring at least 0.7 ft deep; *and*
- A *total* of at least 25 ft in length with a depth of at least 0.7 ft.

Below are examples of the results from six sampling events at a critical riffle site for both the percent contiguous passable width (*Figure 6*) and percent total passable width (*Figure 7*). In these examples, the flow rate (126 cfs) associated with Criterion 1 (Percent Contiguous; *Figure 6*) is the same as Criterion 2 (Percent Total; *Figure 7*), and is therefore identified as the stream flow rate for passage of adult steelhead through the critical riffle site.

Note: If the stream flow rates differ between the two criteria, the higher of the two stream flow rates shall be identified as the stream flow rate for passage through the critical riffle.

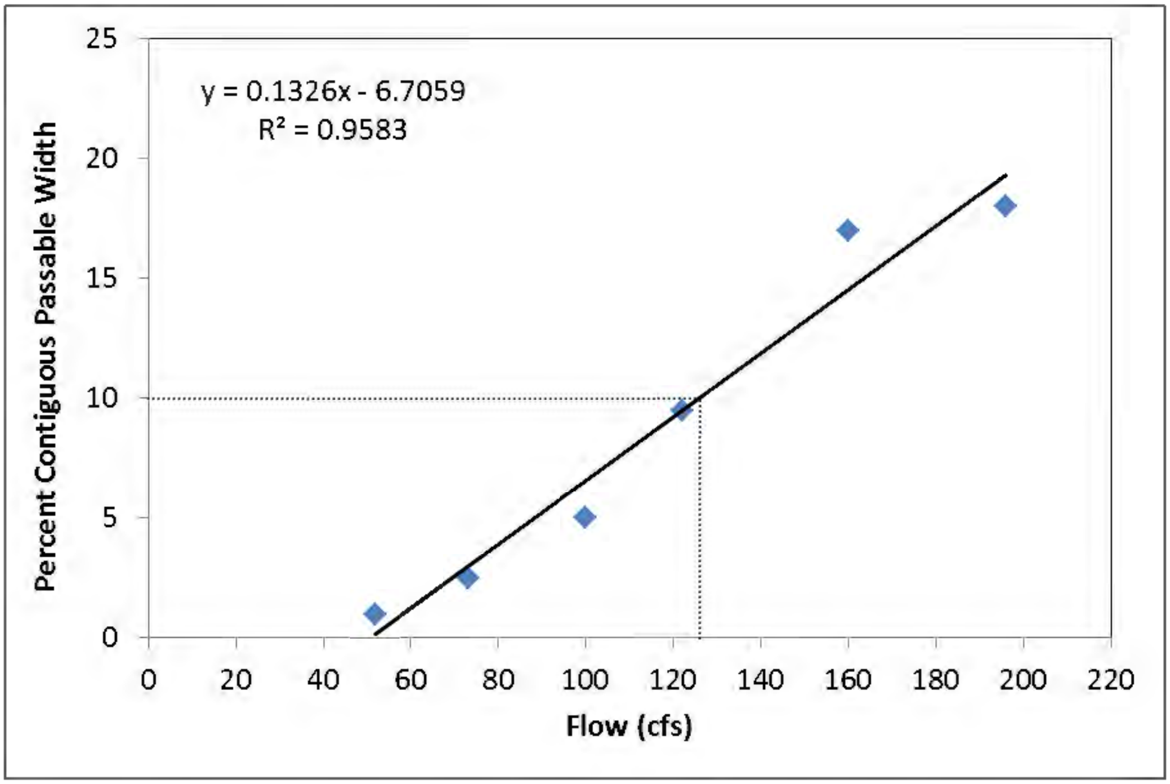

Figure 6. An example of the relationship between river flow (cfs) and percent contiguous passable width for adult steelhead passage. The dashed line represents the flow meeting the 10 percent contiguous passable criteria.

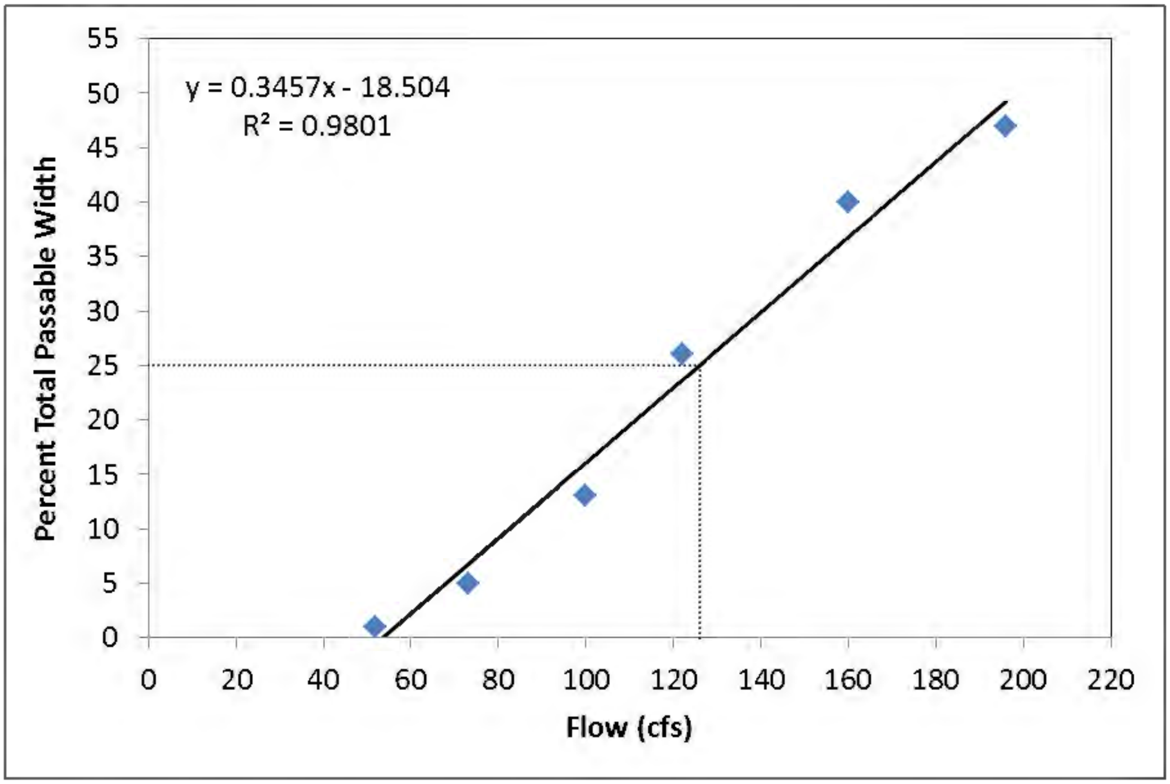

Figure 7. An example of the relationship between river flow (cfs) and percent total passable width for adult steelhead passage. The dashed line represents the flow meeting the 25 percent total passable criteria.

3.5 Considerations for Application of Flows for Salmonid Passage

Several important factors should be taken into consideration during the development and application of flows for passage of salmon and trout. Consider the target species and their life history tactics, with careful attention to approximate time frames for migration and emigration (movement).

Adult salmonids are dependent upon their ability to migrate to spawning habitats at appropriate times that coincide with their life history traits. If adult salmonids are delayed or are unable to reach spawning habitats, the spawning population could be impacted, leading to reduced egg and fry production. Juvenile salmonids are dependent upon their ability to migrate from freshwater riverine habitats (including lagoons) to the ocean. Juvenile salmonids that may rear in freshwater riverine and lagoon habitats (i.e., steelhead) for 1-3 years are dependent upon their ability to access successful rearing habitats in the low-flow summer months. This rearing habitat must have adequate flow (depth and velocity), food, water quality (temperature), and escape cover from predators.

Glossary

Channel Morphology	The dimension (e.g., width, depth), shape and pattern (e.g., sinuous, meandering, straight) of a stream channel.
Critical Riffles	Riffle habitats that may be particularly sensitive to changes in stream flow due to shallow water depth. Critical riffles may prevent adult salmon and steelhead passage to and from spawning areas and/or may prevent movement of rearing juvenile salmonids between adequate summer rearing habitats.
Discharge	The volume rate of water flow transported through a given cross-sectional area. The units that are typically used to express discharge include ft^3/s (cubic feet per second) and m^3/s (cubic meters per second).
Exceedance Probability	The probability that a certain flow value is going to be exceeded.
Thalweg	The lowest line of elevation along the length of a streambed, defining its deepest channel.
Toe of bank	The break in slope at the foot of a streambank where the bank meets the streambed.

References

California Department of Fish and Wildlife (CDFW). 2013a, Standard Operating Procedure for Flow Duration Analysis in California. Department of Fish and Wildlife Instream Flow Program Standard Operating Procedure CDFW-IFP-005, 16 pp. Available at: https://www.wildlife.ca.gov/Conservation/Watersheds/Instream-Flow/SOP

California Department of Fish and Wildlife (CDFW). 2013b. Standard Operating Procedure for Discharge Measurements in Wadeable Streams in California. California Department of Fish and Wildlife Instream Flow Program Standard Operating Procedure CDFW-IFP-002, 24 pp. Available at: https://www.wildlife.ca.gov/Conservation/Watersheds/Instream-Flow/SOP

Flosi, G., S. Downie, J. Hopelain, M. Bird, R. Coey, and B. Collins. 2010. California Salmonid Stream Habitat Restoration Manual, 4th ed. California Department of Fish and Game. Available at: http://www.dfg.ca.gov/fish/Resources/HabitatManual.asp

Hardin, T. 2012. Personal communication with Tim Hardin, Instream Flow Specialist, Oregon Department of Fish and Wildlife. July 2, 2012.

R2 Resource Consultants, 2008. Appendix G: Approach for Assessing Effects of Policy Element Alternatives on Upstream Passage and Spawning Habitat Availability. Administrative Draft prepared for the California State Water Resources Control Board, Division of Water Rights as part of the North Coast Instream Flow Policy: Scientific Basis and Development of Alternatives Protecting Anadromous Salmonids. March 14, 2008. Accessed from: http://www.swrcb.ca.gov/waterrights/water_issues/programs/instream_flows.

Thompson, K.E. 1972. Determining Streamflows for Fish Life. pp. 31-50 in Proceedings of the Instream Flow Requirement Workshop. Pacific N.W. River Basins Commission. Portland, OR.