Hatchery Reform in Washington State: Principles and Emerging Issues

Hatcheries support nearly all major fisheries for Pacific salmon (*Oncorhynchus* spp.) and steelhead (*O. mykiss*) in the Pacific Northwest. However, hatcheries have been a major source of controversy for over 30 years. The Hatchery Scientific Review Group (HSRG) was tasked by Congress to identify solutions to well-known problems so hatcheries could better meet their goals of supporting sustainable fisheries and assisting with the conservation of natural populations. We reviewed over 100 facilities and 200 programs and identified three principles of hatchery reform: (1) goals for each program must be explicitly stated in terms of desired benefits and purposes; (2) programs must be scientifically defensible; and (3) hatchery programs must respond adaptively to new information. We also identified several emerging issues critical to the success of hatcheries. We concluded that hatcheries must operate in new modes with increased scientific oversight and that they cannot meet their goals without healthy habitats and self-sustaining, naturally-spawning populations.

Introduction

An extensive hatchery system for Pacific salmon (*Oncorhynchus* spp.) and steelhead (*anadromous* *O. mykiss*) has developed over the past 100 years in the Pacific Northwest to mitigate for the effects of overfishing, logging, agriculture, hydropower, urbanization, and associated losses of freshwater salmon habitats (Lichatowich 1999). The Washington Department of Fish and Wildlife (WDFW), several Native American tribes, and the U.S. Fish and Wildlife Service operate more than 100 hatchery facilities within Puget Sound and the Pacific coast of Washington state. These hatcheries release more than 100 million juvenile salmon and steelhead each year and contribute approximately 70% of all salmon harvested in Puget Sound (WDFW 1997). The state of Washington and the treaty tribes of western Washington jointly manage fishery resources in western Washington and are collectively referred to as “co-managers.”

A large number of naturally-spawning populations of Pacific salmon and steelhead are currently listed as threatened or endangered under the U.S. Endangered Species Act (www.nwr.noaa.gov/1salmon/salmon/esafractlist.htm). Genetic and ecological interactions with hatchery-origin fish are often cited as one cause of the decline of naturally spawning populations (Waples 1991; Hilborn 1992; Levin et al. 2001). However, the biological effects of hatcheries on natural populations are the subject of much scientific uncertainty and controversy (Busack and Currens 1995; Campton 1995; Brannon et al. 2004). The potential use of hatcheries to help recover natural populations also presents many scientific uncertainties (Hedrick et al. 2000; ISAB 2002; Flagg et al. 2004).

To resolve conflicts between hatcheries and the need for both sustained fisheries and conservation of Pacific salmon and steelhead in Washington state, the U.S. Congress funded the Western Washington Hatchery Reform Project beginning in FY2000 (www.hatcheryreform.org). This project was motivated by recent ESA listings and the important economic and cultural roles of salmon and steelhead fisheries in Puget Sound and coastal Washington. The goal of the project is to determine how hatcheries can best support sustainable fisheries while, at the same time, assist with the conservation and recovery of naturally spawning populations. The project represents a systematic review of all hatchery programs in western Washington, excluding the Columbia River. Congress provided funding to: (a) establish and support an independent panel of scientists to review all hatchery programs and guide the reform process, (b) support a competitive research grants program to address scientific uncertainties associated with hatcheries, (c) support state and tribal efforts to implement hatchery reforms, and (d) facilitate project management by a non-government organization. The independent science panel was established in early 2000 under the name Hatchery Scientific Review Group (HSRG).
Here, we summarize major conclusions resulting from our reviews of hatchery programs during the past four years. Details of the review process have been described elsewhere (Blankenship and Daniels 2004; Blankenship and Kern 2004) but are briefly summarized below (see also www.hatchery.org).

**The Hatchery Review Process**

We visited over 100 facilities and evaluated over 200 hatchery programs encompassing 10 major regions in western Washington (Figure 1). Early in the project, we concluded that hatchery programs must be evaluated separately within each region, particularly in the context of the specific ecosystems and watersheds in which they operate; that is, hatcheries must be considered part of the ecosystem in terms of biomass inputs, biomass outputs, predation and competition effects of released fish, effluents, etc. Moreover, each region has unique attributes (e.g., Seattle versus Olympic National Park) that precluded a single, generalized review.

We evaluated the benefits and risks of each hatchery program within each region relative to: (1) the purpose and goals of the hatchery program where goals focused primarily on harvest and conservation but included other goals such as research, education and outreach, and social/cultural needs of the western Washington tribes; (2) the biological significance of every hatchery and natural stock within the region relative to other conspecific stocks (see Box 1); (3) the population viability of every hatchery and natural stock (see Box 2); and (4) the current and predicted future status of the habitats on which each stock depends. In addition, salmon biologists and fishery managers, representing state and federal agencies and many tribes, provided us with their short-term (15 years) and long-term (50 years) goals (or predictions) for the biological significance and viability of each stock and the habitats on which those stocks depend. Hence, our evaluations did not simply consider current conditions but predicted, future conditions as well.

Our reviews produced over 1,000 program-specific recommendations and 18 system-wide recommendations for Puget Sound and coastal Washington (Table 1). We are now entering an implementation phase. For example, our reviews led to an $8 million appropriation by the state legislature in FY2005 to initiate implementation of our recommendations. Additional funds will be needed in future years, and ultimately some level of prioritization will be necessary. To assist with prioritizing and implementing our recommendations, we have established three principles of hatchery reform as guidelines.

**Three Principles of Hatchery Reform**

**Principle 1: Every hatchery program must have well-defined goals in terms of desired benefits and purpose.**

Well-defined goals provide both targets and measures for success. During our reviews, the goals for many hatchery programs were often not stated clearly or understood by the biologists and managers responsible for the program. For example, the number of juvenile fish released annually was often cited as the goal of a hatchery program. In contrast, we believe the goals for each hatchery program must reflect its intended purpose and desired benefits. The two primary benefits of hatcheries are harvest and conservation, but benefits can include research and education. Wherever possible, goals should be quantified.

Hatcheries should operate as part of an integrated strategy that includes short-term and long-term goals for habitat and harvest. Goals should be related to measures of success, including: (a) the desired number of hatchery-origin fish to be harvested each year so that constraints on the size of the hatchery program can be established, (b) the number of fish returning to a hatchery or spawning naturally in a
Each population or stock was assigned a total score ranging from 3 to 14 according to the following scoring system.

1) What is the genetic origin of the population or stock? (possible scores = 1–5)
   a) Native population. Score = 5.
   b) Genetically admixed population between native and introduced populations.
      i) > 50% native genes? Score = 4.
      ii) < 50% native genes? Score = 3.
   c) Reintroduced population: species occurred historically in watershed, was extirpated, but stock transfers re-established species in watershed. Score = 2.
   d) Introduced population: species was historically absent from watershed. Score = 1.

2) How unique are the biological characters (e.g., life history, physiology, morphology, behavior, disease resistance, etc.) of the stock and to what extent are they considered irreplaceable attributes? (possible scores = 1–5)
   a) Population has unique, irreplaceable biological attributes that are not shared with other stocks/populations within the same Genetic Diversity. Score = 5.
   b) Population has no unique biological attributes, but shares some unique attributes with other stocks/populations within the GDU not shared with other GDUs. Score = 3.
   c) Population has no unique biological attributes that are not shared with other stocks/populations in other GDUs. Score = 1.

3) To what extent is the population or stock part of a larger subdivided population structure or metapopulation? (possible scores = 3–7)
   a) Number of distinct spawning aggregations (e.g., tributaries) within the stock or population under consideration
      i) Number of spawning aggregations < 5. Score = 2.
      ii) Number of spawning aggregations > 5. Score = 1.
   b) Total number of populations or stocks within the GDU.
      i) Number of populations/stocks within GDU < 5. Score = 3.
      ii) Number of populations/stocks within GDU > 5. Score = 1.
   c) What is the viability of other populations or stocks within the same GDU (see Box 2)?
      i) Mean viability = “high.” Score = 1.
      ii) Mean viability = “medium.” Score = 2.
      iii) Mean viability = “low.” Score = 3.

Sum of scores and ratings to assess the biological significance of a population or stock:
14–17: Biological significance = High.
9–13: Biological significance = Medium.
5–8: Biological significance = Low.

1 The Washington Department of Fish and Wildlife defines a GDU as follows: A genetic diversity unit (GDU) is a group of genetically similar stocks that is genetically distinct from other groups. The Stocks typically exhibit similar life histories and occupy ecologically, geographically, and geologically similar habitats. A GDU may consist of a single stock (Busack and Shaklee 1995).

Box 1. Criteria for assessing the biological significance of a population or stock within the Puget Sound and coastal regions of Washington state.
watershed (i.e., escapement), (c) the expected results of scientific research, and (d) the educational benefits to be derived from outreach.

Principle 2: Hatchery programs must be scientifically defensible.

Hatchery programs and operations must be consistent with stated goals, and they must be defensible scientifically. Hatchery programs are often not consistent with goals or the best available scientific information (e.g., Campton 2004). Once the goals for a program are established, the scientific rationale for the design and operation of the program must be explicitly stated and understood by all personnel. These requirements may necessitate a written, comprehensive management plan for every hatchery program. Scientific oversight and peer review should be integral components of every hatchery program.

Every hatchery program provides research opportunities. Indeed, one of the recognized benefits of salmon hatcheries is the opportunity to advance scientific knowledge. For example, every hatchery program creates research opportunities related to the effects of culture on the biology of the propagated species and the effects of the released species on aquatic ecosystems. Partnerships between hatchery staffs and scientists should be encouraged. These research opportunities should be exploited wherever possible.

Every hatchery program needs to have operational guidelines and standard operating procedures (e.g., selection of adults for broodstock, spawning protocols, feeding protocols, etc.) that are scientifically defensible. These guidelines should include decision-making pathways for dealing with unexpected contingencies.

Table 1. Principles for hatchery management and system-wide recommendations developed by the Hatchery Scientific Review Group.

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<thead>
<tr>
<th>Principle 1: Well-defined goals:</th>
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<tr>
<td>• Set goals for all stocks and manage hatchery programs on a regional scale</td>
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<td>• Measure success in terms of contribution to harvest, conservation, and other goals</td>
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<td>• Have clear goals for educational programs</td>
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<th>Principle 2: Scientific Defensibility:</th>
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<td>• Operate hatchery programs within the context of their ecosystems</td>
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<td>• Operate hatchery programs as either genetically integrated or segregated relative to naturally-spawning populations</td>
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<td>• Size hatchery programs consistent with stock goals</td>
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<td>• Consider both freshwater and marine carrying capacity in sizing hatchery programs</td>
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<td>• Ensure productive habitat for hatchery programs</td>
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<td>• Emphasize quality, not quantity, in fish releases</td>
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<td>• Use in-basin rearing and locally-adapted broodstocks</td>
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<td>• Select adults randomly throughout the natural period of adult return</td>
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<tr>
<td>• Use genetically-benign spawning protocols that maximize effective population size and minimize potential artificial or domestication selection under hatchery conditions</td>
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<tr>
<td>• Reduce risks associated with outplanting and net pen releases</td>
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<tr>
<td>• Develop a system of wild steelhead management zones (a special case)</td>
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<td>• Use hatchery salmon carcasses for nutrification of freshwater ecosystems, while reducing associated fish health risks</td>
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<th>Principle 3: Informed Decision Making:</th>
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<td>• Adapтивely manage hatchery programs</td>
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<td>• Incorporate flexibility into hatchery design and operation</td>
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<td>• Evaluate hatchery programs regularly to ensure accountability for success</td>
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Principle 3: Hatchery programs must respond adaptively to new information.

Measuring Success and Accountability. Scientific monitoring and evaluation (M&E) of hatchery programs needs to be expanded to determine whether hatcheries are achieving their goals. If conservation is the goal, a major effort should be made to obtain a census of marked hatchery fish spawning with their natural counterparts and to estimate the impact of hatchery fish on the fitness of the natural component. If harvest is a goal, the focus should be on the contribution of hatchery-origin fish to fisheries and the potential impacts of hatchery fish on natural spawners. M&E should assess smolt-to-adult survival, return rates of adults, contribution of adults to harvest and natural spawning, the proportion of naturally-spawning fish composed of hatchery-origin adults, stray rates of adults to non-target watersheds, and life history traits related to fitness. Where possible, M&E should include assessments of genetic and ecological interactions (e.g., interbreeding, competition, predation, and reproductive success) between hatchery- and natural-origin fish. Biologists should also monitor life history, morphological, and other traits related to fitness because of the potential domestication effects of hatcheries. Most importantly, centralized and standardized databases need to be developed for collating, storing, and retrieving data. Results need to be evaluated annually to allow programmatic adjustments.

In most cases, the existing hatchery staff may not necessarily have the expertise to also serve as M&E biologists. Hence, a hatchery evaluation team, including perhaps the manager of each facility, may need to be developed for a particular region. Indeed, biologists responsible for collecting and analyzing...
data, including the generation of annual reports and publications, should be considered an integral component of every hatchery, equal in importance to fish health specialists who monitor pathogenic loads and help reduce the risk of disease outbreaks. However, it should be recognized that the statistical power to detect undesirable effects, particularly on fitness-related traits, may be very low, so the absence of a detectable effect should not be used as a justification to violate a scientifically-defensible guideline (e.g., as described by Campton 2004). In this context, policies and operational guidelines need to be established and followed accordingly. Moreover, those policies and guidelines need to be subject to change in response to new information generated from M&E activities.

Adaptive Management. Hatcheries need to be flexible and managed adaptively. Many scientific uncertainties are associated with salmon hatcheries. Hatchery programs and facilities must respond to new goals, new scientific information, and changes in the status of natural stocks and habitat. A structured adaptive management program is necessary for the success of hatcheries. Institutional resistance to programmatic flexibility and change needs to be overcome.

We should also note that the three basic principles of hatchery reform are equally applicable to all hatchery programs regardless of whether the purpose of the program is to provide fish for harvest, to assist with conservation of natural populations and their indigenous genetic resources (e.g. Flagg et al. 2004), or both (e.g., Olson et al. 2004). The details of an individual program will be case-specific for both harvest-oriented and conservation-oriented programs, but the underlying principles and operational guidelines will be similar. However, conservation hatchery programs designed to prevent extirpation of an imperiled population do present some special challenges, and those challenges and special considerations are described in detail elsewhere (Flagg et al. 2004).

Emerging Issues and Concepts

A number of key issues emerged during our review process. Here, we summarize our conclusions and recommendations for some of those key issues. More detailed analyses and descriptions are available at www.hatcheryreform.org.

Genetic Integration vs. Segregation of Hatchery Broodstocks Relative to Natural Populations

Adult spawners for broodstock are usually obtained from returnees back to a hatchery or from trapped adults diverted by a barrier weir. Whether such adults are of natural- or hatchery-origin was usually unknown. In short, hatcheries have historically had no formal genetic management plan or strategy for their component broodstocks.

Consequently, the first step towards “hatchery reform” is to develop a detailed genetic management plan or strategy for every hatchery broodstock. Managers have two options: (1) manage a hatchery broodstock as a reproductively distinct population that is genetically segregated from naturally spawning populations, or (2) manage a hatchery broodstock as a genetically integrated component of an existing natural population (Figures 2,3). Each of these broodstock strategies leads directly to a different set of operational guidelines. Genetically segregated broodstocks are generally derived strictly from hatchery-origin adults returning to the hatchery each year. Conversely, genetically integrated broodstocks systematically include a prescribed proportion of natural-origin fish in the broodstock each year to maintain genetic integration with a natural population. One goal of integrated hatchery programs is to minimize the genetic effects of domestication by allowing selection pressures in the natural environment to drive the genetic constitution of hatchery-origin fish and the mean fitness of the population as a whole. In contrast, segregated hatchery programs create a genetically distinct, hatchery-adapted population.

Figure 2. Schematic diagram of a genetically integrated hatchery program. The integrated approach treats hatchery and natural-origin fish as two components of a common gene pool, where \( p_b \) is the proportion of the hatchery broodstock composed of natural-origin fish, and \( p_s \) is the proportion of natural spawners in the watershed composed of hatchery-origin fish. Natural spawning by hatchery-origin fish from a properly integrated program (\( p_b > p_s \)) presents a lower genetic risk than segregated programs (\( p_b = 0 \)) for the same value of \( p_s \). For many hatcheries and geographic locations, the value of \( p_s \) cannot be directly controlled, and the mean value of \( p_s \) may thus be a deciding factor for determining whether a segregated or integrated program is most appropriate for a particular facility and geographic location where fish are released. Regardless of program type, \( p_s \) should be minimized.

Figure 3. Schematic diagram of a genetically segregated program. The segregated approach treats hatchery fish as a distinct population or isolated gene pool. Natural spawning by hatchery-origin fish (\( p_s > 0 \)) from a genetically-segregated program can pose a high risk to natural populations if those hatchery-origin fish are able to reproduce successfully.
The goals of genetic integration can be achieved only if the rate of gene flow from the natural environment to the hatchery environment exceeds the reverse rate of gene flow (e.g., Ford 2002). These gene flow parameters are difficult to estimate directly but can be approximated by the proportion of the hatchery broodstock composed of natural-origin fish ($p_b$) and by the proportion of natural spawners in the watershed composed of hatchery-origin fish ($p_h$), with the goal $p_h > p_b$ (Figure 2). The mean fitness ($f$) of an integrated population in the natural environment, relative to a natural population by itself, will largely be determined by the relationship $f = p_b / (p_h + p_b)$ (Lynch and O’Hely 2001; Ford 2002). The parameter $f$ also equals the proportion of time that genes are transmitted from parents to offspring in the natural environment versus the hatchery environment. At a minimum, we recommend $p_h > 0.1$ for integrated programs to help overcome the effects of genetic drift and unknown amounts of domestication selection in the hatchery environment, even when $p_b$ may be near zero (see Ford 2002). In most cases, $p_h$ will not equal zero because of logistic difficulties controlling the natural spawning of hatchery-origin adults. Consequently, one major motivation for an integrated hatchery program is to substantially reduce the genetic risks of hatchery-fish spawning naturally relative to genetic and ecological risks imposed by hatchery-origin fish from a genetically segregated broodstock. We are currently developing mathematical models to address these concepts further.

An integrated hatchery program requires, as a long-term goal, a self-sustaining, naturally-spawning population capable of providing adult fish for broodstock each year. Integration thus requires suitable natural habitat capable of sustaining a natural population. Under this strategy, an integrated hatchery does not replace habitat but adds to existing habitat. An implicit goal of an integrated program is to demographically increase the abundance of a natural population while minimizing the genetic effects of artificial propagation. This demographic increase in abundance could simply be for the purpose of supporting harvest while minimizing genetic risks to natural populations or it could reflect a conservation component of the program itself. Thus, integrated programs can have both harvest and conservation goals (e.g., Olson et al. 2004). However, the size of an integrated hatchery program will necessarily be limited by the habitat available to the natural populations with which it is integrated and by the ability of the hatchery program to restrain natural spawning by hatchery-origin adults.

Segregated hatchery populations will diverge genetically from naturally spawning populations over time because of founder effects, genetic drift, and domestication selection in the hatchery environment. Such changes may be intentional (e.g., via selective breeding) to maximize benefits or the operational efficiency of a hatchery program. However, natural spawning by hatchery-origin fish can have a significant genetic influence on natural populations after several generations when $p_s$ approaches 10% (Ford 2002); consequently, we recommend that hatchery-origin spawners from genetically segregated programs represent < 5% of the natural spawners as an upper-limit guideline (Figure 3). Segregated programs are, thus, most appropriate when harvest is the principal purpose of the program and the probability of hatchery-origin adults spawning naturally and reproducing successfully is very low. On the other hand, hatchery-origin fish from a segregated broodstock could support a conservation goal if fish were “outplanted” into areas where a natural population was extirpated. Hence, both integrated and segregated programs can have both harvest and conservation goals depending on the specific situation.

The development and management of genetically integrated or segregated hatchery broodstocks clearly requires a mechanism to distinguish natural and hatchery-origin adult fish via intrinsic and extrinsic marks and tags (e.g., otolith marks, DNA profiles, coded-wire tags, fin marks, etc.). Such a distinction is required during the selection of adults for broodstock and for monitoring the potential genetic contribution of hatchery-origin fish to natural reproduction. Moreover, this distinction is required also for monitoring harvest impacts on both hatchery-propagated and natural populations. Hatchery reform cannot occur independently of “harvest reform” and habitat improvements. All three components must be managed together to achieve long-term and sustainable conservation and harvest goals for Pacific salmon and steelhead resources.

During our HSRG reviews, the co-managers identified the intent of every hatchery program as either integrated or segregated. The breakdown between integrated and segregated programs identified by the co-managers was 54% versus 46% respectively. They also needed to identify the purpose of the program as either harvest, conservation, research, and/or education. We then provided recommendations for achieving those collective goals. In some cases, we recommended that they change the intent of their program from an integrated program to a segregated program, or vice versa, if their intended strategy was inconsistent with the capability of the habitat, stray rates from the hatchery, or other factors beyond their control.
immediate control. Simply establishing these hatchery broodstock strategies for each hatchery program was a major step towards satisfying Principle 2 of hatchery reform.

**Marine Carrying Capacity**

Trends in the carrying capacity of the marine environments must be considered for determining the number of fish released from a hatchery. Until recently, marine ecosystems were believed to be stable, internally regulated, and largely deterministic. The current view is that these systems are dynamic with much environmental stochasticity and ecological uncertainty (Mahnken et al. 1998; Francis 2002).

Based on earlier assumptions that marine carrying capacity was unlimited or had not yet been reached, the goal of increasing fisheries was pursued by building more hatcheries and releasing more fish. As a result, the number of juvenile salmon released from Pacific Northwest hatcheries increased substantially after the early 1960s (Mahnken et al. 1998). Equally large increases occurred in Japan and Alaska (NPAFC 2003; Smoker and Heard in press). However, during the same period, the mean length and weight of returning adults decreased and their mean age at maturity increased for most stocks of Pacific salmon (Kaeriyama and Urawa 1992; Rogers and Ruggerone 1993; Bigler et al. 1996). In general, between the mid 1970s and the late 1990s, the marine survival rates of the great majority of Northeast Pacific salmon stocks north of 54°40' N increased rapidly while southern hatchery stocks exhibited decreasing marine survival (Hilborn and Eggers 2000; Wertheimer et al. 2001; Wertheimer et al. 2004). Decreasing smolt-to-adult survival of Southern North American hatchery stocks during the late 1970s and late 1980s only motivated co-managers to release more hatchery fish to compensate for reduced fisheries catch.

The ocean ecosystem in which Pacific salmon reside is dynamic, changing seasonally and on longer decadal scales. El Niños and the Pacific Decadal Oscillation (PDO) are examples of short- and long-term oscillations in the ocean environment that can affect Pacific salmon survival and abundance. At present, it will be difficult to manage Pacific salmon hatcheries on time scales as short as the El Niño owing to the unpredictability of onset of the event. However, for decadal time scales on the order of the PDO, it may be possible to modulate hatchery production to accommodate gradually changing ocean productivity. Agencies need to engage in dialogue to address managing hatchery production during these longer, more predictable periods of natural oscillations in Pacific ecosystem productivity.

**Smolt Quality**

The quality of hatchery-origin smolts has often been described in terms of mean size, numbers, or condition index of fish produced, and whether they met a pre-determined time window for smoltification and release. Fish size at release was used almost invariably as a surrogate for fish quality. Although the scientific literature provides physiological, morphological, and behavioral definitions of a quality smolt, those definitions were rarely used by hatchery personnel. Moreover, the biological relationship between those measures and parameters associated with returning adults (e.g., age class structure, mean size at maturity, etc.) was rarely investigated. With the advent of coded-wire tags, most studies to identify relationships between smolt quality and adult returns were aimed at manipulating the mean size and release time of juveniles with the goal of maximizing smolt-to-adult survivals. In addition, some researchers investigated the role of nutrition (proximate composition, constituent quality, etc.) for maximizing adult returns.

We now recognize that smolt quality includes morphological, physiological, and behavioral characteristics that embody the rate and completeness of the parr-smolt transformation. A quality smolt is defined as a metamorphosed, anadromous salmonid that exhibits rapid downstream migration, increased hypo-osmoregulatory capability, sustained growth in the ocean, and high survival to adulthood. Smolt quality is not an absolute concept; it has to be evaluated in the context of program goals. If the primary goal is conservation, then an analysis of smolt quality needs to consider how hatchery smolts compare in life history traits to wild fish. Mimicking the growth pattern, size, and out-migration timing of natural fish has been shown to have the potential to produce a higher quality hatchery smolt with greater smolt-to-adult survivals (Larsen et al. 2001; Maynard et al. 2004).

As an operational guideline, we recommend expanding the definition of smolt quality to include additional physiological, morphological, and behavioral measurements taken throughout the culture cycle. Examples include gill Na-K ATPase enzyme activity, blood concentrations of thyroid hormones, growth hormone, insulin, insulin-like growth factor, and body lipid levels. Such an approach will require increased physiological monitoring of juveniles prior to release. One simple measure of physiological smolt development is the rate of change in growth rate
Outplanting and Remote Releases.

The vast majority of salmon hatcheries in the Pacific Northwest operate largely as adult spawning and juvenile rearing facilities. The standard method of propagation is to release smolts into stream areas where returning adults can be recaptured for broodstock. In practice, though, smolts are often released from sites where adult collection facilities do not exist, primarily to support fisheries in off-site areas. In many situations, smolts are transported by hatchery truck into other watersheds, sometimes over relatively large distances (e.g., >100 km) prior to release. Adults generally return to areas where they were released as smolts, not where they were reared.

Releasing smolts into streams geographically removed from a hatchery or adult collection facility, primarily for the purpose of supporting fisheries in specific streams or areas, is commonly called “outplanting.” Steelhead programs in Washington State outplant smolts into a large number of small streams to support recreational fisheries where no hatchery facilities exist. Salmon smolts are also released remotely from floating net pens in marine areas where a targeted fishery on returning adults is desired. Outplanting, as defined above, should not be confused with supplementation which, in the Pacific Northwest, refers specifically to the deliberate release of fish in areas where managers explicitly desire returning adult fish to spawn and naturally reproduce as a mechanism to increase the abundance of natural-origin adults one generation later. In general, natural spawning of hatchery-origin fish is not a goal of “outplanting.”

A common feature of outplanting and net pen programs is the absence of facilities to trap returning adults that escape target fisheries. Non-harvested adults can then spawn in streams far-removed from the source hatchery or geographic location where their parents were trapped for broodstock (e.g., Pascual and Quinn 1994). Moreover, stray rates of fish released off-station are generally greater than those of fish released directly from hatcheries (Quinn 1993). Indeed, tagging and genetic studies have shown that outplanting and net pen programs promote stray rates that far exceed natural levels (Candy and Beacham 2000; Mackey et al. 2001). Many studies have further indicated a genetic component to homing such that non-native fish and their progeny stray at higher rates than identically-reared native fish (Bams 1976; McIsaac and Quinn 1988; Pascual et al. 1995; but see Smoker and Thrower 1995; Gilk et al. 2004). We concluded that outplanting and net pen releases pose significant, and potentially unacceptable, genetic risks to natural populations and recommended several measures to reduce risks associated with outplanting (HSRG 2004). These recommendations include reductions in the number of fish released from saltwater net pens, removal of all or nearly all hatchery returnees in concentrated fisheries, the construction of juvenile acclimation and adult recapture facilities, and the potential establishment of wild salmon and steelhead management zones.

Predation on Natural-Origin Salmonids.

Concern is often expressed about the potential for hatchery-reared salmon and steelhead to prey on natural-origin juvenile Pacific salmonids, particularly regarding the impact of predation on threatened or endangered populations (Lichatowich 1999; Levin et al. 2001; see also Sholes and Hallock 1979; Beauchamp 1990; Hawkins and Tipping 1999 for the potential of predation). Predation on natural-origin fish is most likely in the freshwater environment, where potential salmonid predators are concentrated and exposed to large numbers of prey in a relatively small area. There is little evidence that natural-origin salmonids are preyed on by hatchery-reared salmonids in estuarine, nearshore, or offshore marine environments. However, estimation of the overall predation risk to natural-origin salmonids from hatchery-reared salmonids is complicated because such risks depend on the piscivorous nature of an individual stock and on a number of stochastic factors including migration rates, stream conditions, and spatial and temporal overlap between hatchery- and natural-origin fish.

There is evidence that salmonids are able to prey on fish up to approximately 50% of their body length (e.g., Damsgard 1995; Pearsons and Fritts 1999; Finstad et al. 2001). However, Keeley and Grant (2001) estimated that the mean prey size for 100–200 mm salmonids feeding on fish in streams is normally 13–15% of predator body size. The relative sizes of downstream-migrating smolts or fry of different species of salmonids in Washington suggest that several possible predator/prey combinations are likely to occur (Figure 4).

Natural-origin pink (O. gorbuscha) and chum salmon (O. keta) and ocean-type (i.e., that smolt as subyearling) Chinook salmon (O. tshawytscha) are most likely to be preyed upon by hatchery-reared salmonids in Washington. Hatchery-reared chum, pink, sockeye (O. nerka), and ocean-type Chinook salmon are unlikely to prey on wild salmonids due to their relatively small size at release and/or their non-piscivorous feeding habits. Yearling coho (O. kisutch), stream-type (i.e., that smolt as yearlings) Chinook salmon, and steelhead smolts have the greatest likelihood of preying on wild salmonid fry due to their large relative size at release. Smolts that remain (residualize) in rivers for months or years after release may represent an important predation risk to wild salmon populations.
Hatchery managers have numerous production tools that can minimize predation risk of migratory hatchery fish with their natural-origin counterparts. For example, design of rearing and release protocols allow the hatchery manager the option of producing fully-developed, highly-migratory smolts that rapidly migrate to the marine environment. The relative vulnerability of wild juvenile salmonids to predation in freshwater may depend on the release location of hatchery fish. Hatchery managers will need to conduct case-by-case determinations of predation risk that may require specific research on spatial and temporal overlap of hatchery-reared and wild fish, estimation of specific predation rates, and modeling to determine the potential population effects on wild salmonid populations.

**Stream Nutrification and Fish Health**

Returning adult salmon transfer marine nutrients into terrestrial and freshwater ecosystems, contributing significantly to primary production, riparian vegetation, and even old-growth forests (Kline et al. 1990; Bilby et al. 1996). Those nutrients also contribute significantly to the food sources of juvenile salmonids. For example, in one study, up to 60% of the fixed nitrogen in benthic insects was derived from salmon carcasses (Johnston et al. 1997). Those latter investigators also found that juvenile salmon have higher growth rates in streams where adult salmon spawn than in streams without spawning adults. Hatchery-origin salmon carcasses also increased the density of age 0+ coho salmon and age 0+ and 1+ steelhead in small streams in southwestern Washington (Bilby et al. 1998). Distributing spawned-out salmon carcasses from hatcheries into watersheds can thus confer a positive ecological benefit to stream nutrification and naturally spawning populations.

However, spawned-out salmon carcasses from hatcheries can pose a fish health risk to natural populations if the carcasses are not properly treated or inspected prior to distribution. Pathogenic organisms present in salmon carcasses can be transmitted to other salmonids either through release of those organisms into water or through their direct consumption. It is well established, for example, that bacterial and viral pathogens of salmonids can be transmitted via water or the oral route, and, in the laboratory, challenges by both routes have been used in various studies for establishing infections with a number of microbial fish pathogens (see, for example, Evelyn 1996 for various bacterial fish diseases and Helmick et al. 1995 for infectious hematopoietic necrosis virus infections). In addition, the shedding of bacteria and viruses responsible for these diseases from infected fish has been well documented (see, for example, Mulcahy et al. 1983 and Zhang and Congleton 1994 for infectious hematopoietic necrosis; McKibben and Pascho 1999 for bacterial kidney disease; Rose et al. 1989 and Perez et al. 1996 for furunculosis; Madetoja et al. 2002 for bacterial coldwater disease). Such horizontal transmission is enhanced in hatcheries. Adult salmon trapped for hatchery broodstock are typically held in crowded ponds or raceways for days or weeks prior to spawning, potentially exacerbating the prevalence and transmission of pathogenic organisms among hatchery-spawned adults. In many of those fish, the infectious load can be expected to approach the loads found in fish dying of various infections. In such fish, the infectious load of bacterial cells or virus particles can reach titers of $10^3$ to $10^9$ infectious units per gram of tissue, depending on the tissue (see Evelyn 2001). Consequently, every effort should be made to ensure that hatchery-origin carcasses intended for use in stream nutrification projects undergo some form of post-spawning treatment to minimize the disease risks to natural populations (e.g., deep freezing of the carcasses at $-20^\circ$ C, which kills certain fish pathogens or preferably processing of the carcasses into pellets, such as those studied by Wipfli et al. (2004) would almost certainly be pathogen-free). Failing that, pathogen-free certification of the carcasses prior to distribution for stream nutrification projects would be strongly recommended.

**Conclusions**

Salmon hatcheries have been a major source of controversy in the Pacific Northwest for over 30 years (Lichatowich 1999; Brannon et al. 2004). Several panels of scientists have been assembled in the past to identify and evaluate the biological risks posed by hatcheries on anadromous salmonid resources (e.g., IHOT 1994; NRC 1996; Brannon et

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**Figure 4.** Relative sizes of hatchery-reared salmonid predators (at release) and their potential migratory salmonid prey in Washington, assuming that predators may consume fish up to 50% of their body size.
Virtually all of those previous assessments have focused on the problems and risks posed by hatcheries. However, few of those assessments were tasked with developing scientifically-defensible solutions to the problems they identified.

The HSRG was mandated by Congress to identify potential solutions to widely-recognized problems to ensure that hatcheries contribute to supporting sustainable fisheries while supporting conservation, restoration, and recovery of natural populations. A significant portion of the Puget Sound area is urbanized, and hatcheries provide an essential component to the commercial, tribal, and recreational fisheries of the region. In addition, major habitat restoration programs are in place, or are planned, thus providing significantly improved opportunities for naturally spawning populations in the future.

We focused our review efforts during the past four years on identifying scientific uncertainties and proposing solutions based on the best available science. The need to develop broodstock genetic management plans for every hatchery program with the goal of managing each broodstock as either a genetically-segregated “hatchery population” or as a genetically-integrated component of an existing “natural population” became a fundamental foundation for our recommendations. Both strategies require the ability to distinguish hatchery- and natural-origin adults, both in the hatchery when adults are spawned for broodstock and on the natural spawning grounds, to assess the genetic risks and gene flow rates of hatchery-origin fish to natural populations. Commensurate with these reforms is the need for increased monitoring and evaluation, scientific oversight, and accountability of hatchery operations. In many cases, these reforms will require additional funding by the management agencies, but this investment should be considered a cost of operating hatcheries to ensure benefits are achieved and risks controlled. Ultimately, the success or failure of a hatchery program may be measured by the relative benefits and risks it confers. Although it is beyond the scope of this article to deal with benefit-risk analyses of hatchery operations, readers should consult other essays on this topic as a complement to our presentation here (Busack and Currens 1995; Currens and Busack 2004; Waples and Drake 2004).

Unless all habitat has been irrevocably lost to a dam or other impassible barrier, hatcheries should not be regarded as surrogates for lost habitat. Hatcheries need to operate in scientifically-defensible modes with well-defined goals and substantially increased data collection and evaluation. Hatcheries also need to be flexible and adaptable; that is, they need to operate and be evaluated in the context of both the ecosystem (watersheds) in which the hatcheries occur and other ecosystems and ecological processes on which hatchery-origin fish depend.

Scientific uncertainties associated with hatchery operations are numerous. The science to manage these risks is still inadequate, and some of the risks are still poorly understood (e.g., Currens and Busack 2004). However, one point is clear. Maintaining healthy habitat is critical not only for viable, self-sustaining natural populations, but also to adequately control risks of hatchery programs and realize the benefits of hatcheries to recover populations and sustain healthy harvests in increasingly populated environments. Moreover, the principles and emerging issues we describe here are not only applicable to hatcheries for Pacific salmon and steelhead in western Washington state, they also have direct application to other regions of the Pacific Northwest and to artificial propagation programs for other species worldwide.

Acknowledgements

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References


Prospects for Recovering Endemic Fishes Pursuant to the U.S. Endangered Species Act

If the success of the Endangered Species Act (ESA) is measured by the number of endangered species that have been recovered and delisted, then the act is not very successful. Only 15 species have been delisted because of recovery in the history of the ESA. The Borax Lake chub (Gila boraxobius), an endangered species restricted to an Oregon spring system, is considered to be on the brink of recovery and may warrant future delisting. A panel of scientists was convened to determine consensus regarding the species’ listing status by reviewing: (1) current habitat conditions, (2) implementation of the recovery plan, and (3) applicability of ESA listing factors. Despite substantial progress towards recovery, threats to the species remain, including habitat degradation and the potential introduction of nonnative species. These are problems common to many fishes of highly restricted distribution. Because the Borax Lake chub occurs in a single spring system, the species remains vulnerable to catastrophic loss and requires continuing protection afforded by the ESA. Like many spring-dwelling fishes with a restricted range, recovery of the Borax Lake chub to the point where ESA protection is no longer required is an admirable but largely unobtainable goal. Prevention of extinction rather than delisting is a more appropriate measure of ESA success for such species.

According to Section 2 of the Endangered Species Act of 1973 (ESA), the primary purpose of the act is to stem the tide of human-caused extinctions and to provide a means whereby the ecosystems upon which endangered and threatened species depend may be conserved. The ESA is widely regarded as the most important conservation law in the United States and is viewed as the pinnacle of legislation for protecting wildlife (Bean 1983; Plater 2004). Because of its importance and influence, the ESA has been the cornerstone for a growing number of conservation battles across the country. Conflicts between application of the ESA and land and water development projects have increased because of several factors, but chief among these is the cumulative effects of a growing human population and increasing resource demand coupled with an increasing number of species listed as endangered or threatened.

The 30-year history of the ESA has been characterized by a growing list of protected species, subspecies, and distinct population segments (hereinafter “species”). When the ESA was signed into law by President Nixon in 1973, 119 species received “grandfathered” protection from the earlier Endangered Species Conservation Act of 1969. From 1973 through 2002, an average of nearly 43 species were added each year to the list of endangered and threatened species until a total of 1,262 species (517 animals, 745 plants) were listed in the United States as of 2003 (U.S. Fish and Wildlife Service [USFWS] 2004a).

Over this same timeframe, 37 species have been delisted and subsequently removed from ESA protection. Of these, 15 were delisted because of recovery, 7 because of extinction, and 15 because of new information or taxonomic revision showing their listing was in error (USFWS 2004b). The low number of recovered species is due largely to inadequate protection from a growing array of threats to species and habitats, and because delisting removes the primary regulatory protection available—that is, from the ESA itself (Doremus and Pagel 2001). Indeed, some scientists and legal scholars have questioned whether we are likely to see the recovery of many listed species, and instead have proposed that recovery should be viewed as an aspirational goal rather than a realistic expectation for many listed species (Doremus 2000; Doremus and Pagel 2001). On the other hand, others have encouraged delistings because of recovery for a variety of practical, political, and philosophical reasons (Bender et al. 1998).

The growing list of protected species and increasing human-caused fragmentation and degradation of natural habitats presents a looming conflagration for conservation efforts. As conflicts escalate between the ESA and human development, there are growing efforts to reduce the impact and effectiveness of the ESA. One way to reduce the impact of the ESA is to reduce the number of protected species, either by slowing the number of new species listings and/or increasing

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the number of delistings. For the first 20 years following passage of the ESA, there were only 18 delistings, but since 1993 the rate of delistings has increased. Elements on both sides of the conservation debate have sought to increase the number of delisted species. During the Clinton Administration, Interior Secretary Babbitt believed that the USFWS's ability to delist species because of recovery was a clear indication that the ESA was a success. During this period, USFWS expedited delisting efforts, including development of lists of species that might warrant delisting because of recovery (Bender et al. 1998). As Rohlf (2004) pointed out in a recent review of Section 4 of the ESA, federal agencies have both political incentive and institutional desire to find success in the ESA by pointing to recovered species that may be delisted. Of course, those that oppose the ESA are equally glad to see fewer species protected under the act's provisions, but for different reasons.

Among the species that might warrant contemporary delisting is the Borax Lake chub (*Gila boraxobius*), an endangered species inhabiting a small hot-spring ecosystem in southeastern Oregon. The restricted habitat occupied by the species recently has been acquired by a conservation group and surrounding public land has received additional protections. In 2003, we conducted a review of the conservation status of the Borax Lake chub to develop a scientific consensus regarding the listing status and future conservation needs of that species. The purposes of this article are to report on the results of our evaluation of the Borax Lake chub, discuss implications of our finding for the vulnerability of other species of restricted range, and to provide recommendations for status reviews for endemic species listed pursuant to the ESA. We also offer our opinion regarding appropriate criteria for measuring the success of the ESA itself. Management of endangered and threatened fishes, including their recovery and delisting, are critical topics to fisheries biologists. We hope this article stimulates further debate on measures of success for the ESA and understanding of the appropriate role of delisting.

**Case Study: The Borax Lake Chub**

The Borax Lake chub is endemic to the geothermally-heated waters of Borax Lake and adjacent wetlands in Oregon's Alvord Basin (Williams and Bond 1980). The chub was listed as endangered in 1980 by emergency rule and again as endangered by final rule in 1982 (USFWS 1982). At the time of listings, the primary threats to the species consisted of potential impacts from geothermal energy development and diversion of the lake's outflows by alteration of the shoreline crusts. Although no recovery team ever was formed for this species, a recovery plan was completed in 1987 that called for protection of the Borax Lake ecosystem through acquisition of key private lands, protection of subsurface and surface waters, controls on access, removal of livestock grazing, monitoring, and other recovery actions (USFWS 1987).

The Borax Lake chub exists as a single population that most likely has been maintained within its historic range of natural variability, and an increase in abundance is not a factor in successful recovery. Recovery, in this instance, is based entirely on habitat integrity, including protection of spring aquifers, and the avoidance of nonnative species introductions.

Borax Lake is a spring-fed ecosystem in Oregon's Alvord Desert and, along with surrounding pools and marshes, the sole habitat for the endangered Borax Lake chub.
Numerous recovery measures have been implemented during the past two decades to secure habitat for the species. In 1983, the Bureau of Land Management (BLM) designated the public lands surrounding Borax Lake as an Area of Critical Environmental Concern. The Nature Conservancy (TNC) leased two 160-acre private land parcels, one surrounding Borax Lake and the other immediately to the north, in 1983 and purchased them outright in 1993, thereby bringing all lands designated as critical habitat into public or conservation ownership. With the acquisition by TNC, livestock grazing ceased. Passage of the Steens Mountain Cooperative Management and Protection Act of 2000 withdrew public lands from mineral and geothermal development within a majority of the Alvord Basin, including the Alvord Known Geothermal Resource Area and Borax Lake.

With removal of many of the significant threats facing the Borax Lake chub, the U.S. Fish and Wildlife Service began to examine its feasibility for reclassification (R. White, USFWS, pers. comm.). The Borax Lake chub frequently is cited by USFWS as being “on the brink of recovery” (Motivans and Balis-Larsen 2003) and is rated by that agency as having achieved a relatively high percentage of recovery implementation (51–75%; USFWS 2003). In 2003, two of the authors conducted a status review of the Borax Lake chub to determine whether a change in listing status was warranted and to review future management and monitoring needs for the species (Williams and Macdonald 2003). The status review consisted of four components: (1) review of recovery plan implementation, (2) field investigations at Borax Lake to determine current status of the species and habitat, (3) review of the five listing factors from Section 4 of the Endangered Species Act, and (4) convening of a 16-member scientific panel to review findings from the recovery plan, habitat, and listing factor reviews. The panelists were scientists that had worked previously on the species and its habitat, agency biologists with management responsibility for the species, and other scientists with extensive knowledge of desert spring systems in western North America. Panelists were asked, using their best scientific judgment on issues rather than agency positions, to develop a consensus on listing status, management, and monitoring.

The expert panel concluded that substantial progress has been made towards recovery of the Borax Lake chub, but that despite this progress, threats to the species and ecosystem remain. Results of the status review are summarized in Table 1. Threats that had been eliminated included the alteration of lake shoreline and outflows, livestock grazing, and geothermal energy development on public lands. The primary remaining threats were increasing habitat degradation associated with recreational use and the increasing potential of nonnative species introduction. Exotic goldfish (Carassius auratus) recently have been introduced into Mann Lake just to the north of Borax Lake (Tim Walters, Oregon Department of Fish and Wildlife, pers. comm.). Both recreation and introduced species received minor attention in the 1987 recovery plan. Borax Lake is located in a remote and sparsely-populated area, but one that is increasingly used by a public seeking opportunities for solitude, wildlife observation, and open space. The panel believed that because the range of the Borax Lake chub is restricted to single geologically fragile site, the species is vulnerable to catastrophic loss despite existing protection. The panel also noted the importance of frequent monitoring to detect and move to extirpate

### Table 1. Summary of status review findings for the Borax Lake chub

<table>
<thead>
<tr>
<th>Factor</th>
<th>1982</th>
<th>2003</th>
</tr>
</thead>
<tbody>
<tr>
<td>1. Overutilization for commercial, sporting, scientific or educational purposes</td>
<td>no threats</td>
<td>threats consist of recreational use and potential water development on private lands.</td>
</tr>
<tr>
<td>2. Overutilization for commercial, sporting, scientific or educational purposes</td>
<td>no threats</td>
<td>threats consist of recreational use and potential water development on private lands.</td>
</tr>
<tr>
<td>3. Disease or predation</td>
<td>no threats</td>
<td>no threats for this factor.</td>
</tr>
<tr>
<td>4. Inadequacy of existing regulations</td>
<td>no threats</td>
<td>threats consist of recreational use and potential water development on private lands.</td>
</tr>
<tr>
<td>5. Other natural or manmade factors affecting its continued existence</td>
<td>no threats</td>
<td>threats consist of recreational use and potential water development on private lands.</td>
</tr>
</tbody>
</table>

#### Recovery Plan Implementation

<table>
<thead>
<tr>
<th>Task</th>
<th>Subtask Score</th>
</tr>
</thead>
<tbody>
<tr>
<td>Task 1: Secure land and water rights.</td>
<td>3.7</td>
</tr>
<tr>
<td>Task 2: Restore Lower Borax Lake, small ponds, and intervening marshes.</td>
<td>4.0</td>
</tr>
<tr>
<td>Task 3: Protect Borax Lake ecosystem.</td>
<td>2.7</td>
</tr>
<tr>
<td>Task 4: Monitor status of ecosystem.</td>
<td>2.3</td>
</tr>
<tr>
<td>Task 5: Encourage support of recovery through public awareness.</td>
<td>3.5</td>
</tr>
</tbody>
</table>

#### Field Investigations

Habitat and chub population appeared in good condition and within expected range of variation observed historically. Significant recreational use (off-road vehicle use, camping, disturbance of lake substrates from wading) was noted.

#### Review of 5 Listing Factors

1. Present or threatened destruction, modification, or curtailment of its habitat or range. 1982: threats consisted of chipping of crusts around shoreline, diversion of outflows, development of geothermal resource, and potential development of recreation facility. 2003: threats consist of recreational use and potential water development on private lands. 2003: threats consist of recreational use and potential water development on private lands.

2. Overutilization for commercial, sporting, scientific or educational purposes. 1982, 2003: no threats for this factor.


5. Other natural or manmade factors affecting its continued existence. 1982: no threats for this factor. 2003: because of restricted range, species vulnerable to disturbance event.
introduced species and to be able to act quickly in the face of other new threats. No change in listing status was recommended although the expert panel concluded reclassification from endangered to threatened could be appropriate in the near future depending primarily upon implementation of a regular monitoring program. The panel further concluded that “maintaining the Borax Lake chub on the list of Endangered and Threatened Wildlife and Plants affords the greatest likelihood that sufficient scientific and agency attention will be focused on Borax Lake such that if habitat integrity is compromised, corrective action will be timely enough to save the species.”

**Delisting and Vulnerability of Endemic Fishes**

Given the plethora of possible causes of population endangerment, determining vulnerability of species to extinction events is difficult. Many factors are relevant, including a species’ habitat requirements, population size, and dispersal abilities (Tilman et al. 1994; Driscoll 2004). Furthermore, a search for explanations of status changes in many lesser-known species listed as endangered or threatened often is hindered by our lack of knowledge of their basic life history and habitat requirements. Nonetheless, certain factors common to many endangered species are known to increase the likelihood of their extinction. These factors include small population size (Soule 1983; Gilpin and Soulé 1986), restriction to a small geographic area (Lovejoy et al. 1986), dependence upon a specific rare habitat type (Terborgh 1974), and inability to move away from increasing sources of stress or habitat degradation (Diamond 1975).

Endemic fishes with a highly restricted range are particularly vulnerable to extinction because they occur as a single or low number of populations, depend upon a specific habitat type, and have low tolerance for habitat modification. Endemic fishes may be common in the limited areas where they occur but often have rigid habitat requirements. These endemic species therefore, become highly vulnerable to habitat change or invasion of nonnative species (Minckley and Deacon 1968; Terborgh 1974). The vast majority of recent U.S. extinctions have been in species with restricted ranges, including freshwater mussels of southeastern rivers, and plants and terrestrial invertebrates of Hawaiian forests (Suckling et al. 2004). In their review of western fish conservation, Deacon and Minckley (1991) concluded that the restricted distributions and small population sizes of many spring-dwelling fishes dictated their virtual permanent status as endangered or threatened.

Of the 15 species that have been delisted because of recovery, most are wide-ranging, such as the American alligator and peregrine falcon. Five fishes have been removed from the list of threatened and endangered species, four because of their extinction (Tecopa pupfish [Cyprinodon nevadensis calidus], longjaw cisco [Coregonus alpestris], blue pike [Stizostedion vitreum glaucum], and Amistad gambusia [Gambusia amistadensis]) and one because of taxonomic revision (Umpqua River form of coastal cutthroat trout, Oncorhynchus clarki clarki). No fishes have been delisted because of recovery. Although it is difficult to generalize about the characteristics of listed species that make good candidates for recovery, it seems clear that species with the following suite of characters may more readily respond to recovery efforts: (1) habitat requirements are more general than specific, (2) quality habitat remains within historic range, and (3) existing threat factors, such as overharvest, may be easily regulated. Simply stated, recovered species often faced threats that were easier to address through available regulatory channels (Abbit and Scott 2001). On the other hand, species from specialized habitats and/or smaller ranges may be more vulnerable to loss (Terborgh 1974; Deacon and Minckley 1991). In a review of the conservation status of aquatic species in the Great Basin, Sada and Vinyard (2002) found that declines were greatest in the most narrowly distributed and vulnerable populations. According to their analysis, all extinct taxa and most taxa suffering major declines (68%) had fewer than five small populations.

A report by Bender et al. (1998) listed 22 species considered likely candidates by USFWS for delisting or reclassification because of increased protection, included three fishes: tidewater goby (Eucyclogobius neuberryi), Ash Meadows pupfish (Cyprinodon nevadensis mionectes), and Pahrump poolfish (Euteignorhinchus latos). The tidewater goby is more broadly ranging, but the Ash Meadows pupfish and Pahrump poolfish are both spring-dwelling fishes with restricted ranges that are similar to the Borax Lake Jackrabbit Spring in Ash Meadows, Nevada, provides habitat for the endangered Ash Meadows pupfish and Ash Meadows speckled dace (Rhinchithys osculus nevadensis). Other nearby springs provide habitat for the endangered Devils Hole pupfish (Cyprinodon diabolis) and warm springs pupfish (C. nevadensis pectoralis). Despite designation of the springs as protected areas (Ash Meadows National Wildlife Refuge and disjunct portion of Death Valley National Park), habitats and fishes remain vulnerable.
chub in terms of vulnerability. The Ash Meadows pupfish and three other listed fishes are endemic to springs in the Ash Meadows area but remain vulnerable to catastrophic loss because of introductions of nonnative species and/or modification to subsurface aquifers. These threats persist despite protective management of land around surface spring areas. The recovery plan for Ash Meadows species lists protection of aquifers, eradication of nonnative species, and restoration of natural spring habitats as essential criteria that must be met before fishes should be considered for delisting or reclassification from endangered to threatened (USFWS 1990). Recovery of the Pahrump poolfish is doubtful. This species has been eliminated from its single spring historic habitat but exists as an introduced population on the Desert Wildlife Range. Like the Borax Lake chub, these spring-dwelling desert fishes are likely to need the protection afforded by the ESA in perpetuity.

Current procedures for delisting species pursuant to Section 4 of the ESA are similar to listing. That is, the status of the species is compared to the five listing factors contained in Section 4, and if delisting is believed warranted by USFWS, a proposal is published in the Federal Register notifying the public of the proposed change and seeking public comments. We suggest the panel review conducted for the Borax Lake chub may provide a suitable model to evaluate the ESA status of endemic species, particularly those lacking recovery teams. For species with recovery teams, the team likely could substitute for the expert panel. Regardless, a variety of factors should be reviewed in any delisting process, including the implementation status of any applicable recovery plans and current status of subject populations and habitats, in addition to an analysis of the five listing factors.

Conclusions

The desire to delist species is driven, at least in part, by the belief that recovery of listed species is an indicator of the success of the ESA. But with only 15 taxa delisted because of recovery in the history of the ESA, success as measured by this indicator is poor. More appropriate indicators would include changes in population trends of listed species and the ability of ESA protections to prevent extinction. In its latest biennial report to Congress on recovery of listed species, USFWS (2003) reported that population trends for 39% of listed taxa were either stable or increasing, while 34% were declining, and 24% were uncertain. Pursuant to this indicator, the ESA fares better. If preventing extinction is the criterion, an assessment of the success of the ESA is even more positive, with only 7 taxa delisted because of extinction. One study estimated that based on risk of extinction alone, 192 listed taxa would have been expected to go extinct between 1973 and 1998 (Schwartz 1999). Recent data from the Center for Biological Diversity (Suckling et al. 2004) supports the value of ESA protections in preventing extinction. An analysis of 114 extinctions of U.S. species since the ESA was passed in 1973 found that 81% of extinction events involved taxa that were not protected by the ESA (19% were listed). Suckling and others (2004) believe that removal of procedural delays in listing species pursuant to the ESA and elimination of the listing backlog would have resulted in increased protection that likely would have prevented many extinctions. Additionally, many rare but non-listed species occur with listed species and may receive protection that could be indirectly credited to the ESA. Regardless, the small number of extinctions of listed species suggests strongly that the ESA has been successful in ensuring the continued existence of the taxa it protects.

Delisting of spring-dwelling fishes with restricted ranges should be approached with considerable caution because of their inherent ecological and biological vulnerability and their ability to serve as umbrella species protecting many lesser-known and unlisted organisms. Although the Borax Lake chub has a higher rate of recovery success than many listed species, it appears to an expert panel to be a poor candidate for delisting largely because of its inherent vulnerability as an endemic species dependent on a specialized habitat. Ironically, because the ESA is a strong regulatory law for species and habitat protection, removal of this protection through the delisting process also removes the preeminent tool for maintaining the species in the long run. The naturally restricted range of many endemic fishes makes their recovery to the point of delisting an admirable but largely unobtainable goal.

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References


Moving Forward from Recovery under the U.S. Endangered Species Act to Long-term Conservation of Inland Cutthroat Trout

Introduction

The Endangered Species Act of 1973 (ESA, as amended through the 100th U.S. Congress) is the most important U.S. federal law intended to aid plants and animals that are in danger of extinction (NRC 1995). Key purposes of the ESA are “to provide a means whereby the ecosystems upon which endangered species and threatened species depend may be conserved, [and] to provide a program for the conservation of such endangered species and threatened species” (ESA section 2(b)). "Endangered species" is defined as any species (or subspecies or, for vertebrates, “distinct population segment”) which is in danger of extinction throughout all or a significant portion of its range, whereas “threatened species” is any species which is likely to become an endangered species within the foreseeable future (ESA section 3). The ESA does not define “foreseeable future.”

The ESA's section 4(a)(1) provides the five factors of potential threats to a species that must be examined when determining whether the species is endangered or threatened: “(A) the present or threatened destruction, modification, or curtailment of its [the species'] habitat or range; (B) overutilization for commercial, recreational, scientific, or educational purposes; (C) disease or predation; (D) the inadequacy of existing regulatory mechanisms; and (E) other natural or manmade factors affecting its continued existence.” Listing of the species as endangered or threatened may be judged appropriate due to threats from one or more of those factors. Such determinations must be made solely on the basis of the best available scientific and commercial information, after a review of the species’ status has been conducted and efforts being made to protect and conserve the species have been considered (ESA section 4(b)(1)(A)).

Section 4(f) specifies development and implementation of recovery plans for listed species. Recovery plans, which should be developed within 30 months of species’ listing (Federal Register 59: 34272 [1 July 1994]), must include: “(i) a description of such site-specific management actions as may be necessary to achieve the plan's goal for the conservation and survival of the species; (ii) objective, measurable criteria which, when met, would result in a determination, in accordance with the provisions of this section, that the species be removed from the list; and (iii) estimates of the time required and the cost to carry out those measures needed to achieve the plan's goal and to achieve intermediate steps toward that goal.” Although not specified in the ESA, “recovery” has been defined by the U.S. Fish and Wildlife Service (USFWS 1990), one of two federal agencies responsible for administering the ESA, as “the process by which the decline of a threatened or endangered species is arrested or reversed, and threats to its survival are neutralized, so that its long-term survival in nature can be ensured.” While there has been no evident disagreement that Congress intended for recovery management actions to neutralize threats to listed species, the precise nature of recovery criteria is a matter of continuing debate (e.g., Doremus 2000; Doremus and Pagel 2001; Kline 2001).

Behnke (1992) recognized 13 subspecies of cutthroat trout (Oncorhyncus clarki) native to inland areas of North America. Two of those cutthroat trout are now considered extinct and three of the extant subspecies, i.e., greenback cutthroat trout (O. c. stomias), Lahontan cutthroat trout (O. c. henshawi), and Paiute cutthroat trout (O. c. selenis), have been listed (presently as threatened) for 36 to 38 years under the ESA and its predecessors (see also Petersen 1999; Doremus 2000). We present our opinion on fundamental conflicts involving species’ recovery under the ESA, as now being implemented for the listed cutthroat trout, and offer a new paradigm for moving forward from the present, seemingly intractable recovery process to long-term conservation of the fish.

The Contemporary Recovery Paradigm

In our opinion, the Contemporary Recovery Paradigm for the listed cutthroat trout is characterized by periodic attempts to revise established recovery criteria, emphasis on administrative process rather than progress toward recovery, and antagonism, pessimism, and apathy among recovery-program participants. Perhaps most important, the Contemporary Recovery Paradigm (Figure 1) makes no distinction between recovery of listed cutthroat trout under the ESA and the subspecies' long-term conservation.

For each of the listed cutthroat trout, recovery plans and their implementation programs have been in place and functioning for many years. Nevertheless, for the Lahontan and Paiute cutthroat trout recovery programs, a lack of stable, objective, and measurable recovery criteria has been problematic. Opinions regarding Lahontan cutthroat trout recovery criteria recently became so disparate that one key agency threatened to withdraw from participation in important elements of the recovery...
process (Crawforth 2001). In part, that response resulted from attempts to revise established recovery criteria by emphasizing the establishment of Lahontan cutthroat trout metapopulations, particularly those associated with large, terminal lakes, while reducing the importance of extant, isolated populations in streams. Such metapopulations do not now exist and some biologists believe they are unattainable because, among other things, competing or potentially hybridizing, nonnative fishes cannot be eliminated. Paiute cutthroat trout, a listed species for 38 years, recently had objective recovery criteria added to its recovery plan, after consensus finally had been reached on the extent of the subspecies’ historic range. Only the greenback cutthroat trout recovery plan, first written in 1977, has escaped notable problems related to recovery criteria. Nevertheless, the USFWS (administrator of the ESA as regards inland cutthroat trout) was recently asked to revise those recovery criteria by requiring additional and more-secure populations for recovery than described in the existing recovery plan (BLF 2001; see also Young and Harig 2001). The request, which has not been acted upon by the USFWS, was largely based on emerging theory related to population viability.

Former Interior Secretary Bruce Babbitt has been quoted (Marston 2001) as having said: “The environmentalists’ job is to move the goalpost.” We believe recovery-team members and other persons involved in the recovery process may also engage, wittingly or otherwise, in attempts to “move the [recovery] goalpost.” The ESA (section 4(b)(3)(B)) requires that the process for determining whether a species should be listed as endangered or threatened be completed within 12 months of receipt of the listing petition. In contrast, although the recovery plan should be prepared within 30 months of listing, the ESA does not specify a schedule for implementing the plan or achieving recovery. Consequently, the recovery process may be especially prone to self-perpetuation. That pattern may be exemplified in the current recovery plan for Lahontan cutthroat trout, which states...because species recovery is a dynamic process and recovery plans are based on the best available biological information at the time, the recovery plan should be updated periodically...thereafter, the plan should be reviewed, evaluated, and revised when appropriate tasks are completed, or as new information becomes available.” Thus, although the Lahontan plan provides no clear schedule for recovery, it has a provision for routine plan revision. Another pattern that we have observed is for recovery teams to expand the number of threats as recovery plans are being written or revised. Although those additional threats may be valid and became evident only after the species was listed, a desire to eliminate all plausible threats to the species, regardless of the threat’s importance to the species’ risk of extinction, may also be a factor. In any case, we believe that recovery for each of the listed cutthroat trout has become a seemingly self-perpetuating process whose endpoint remains vague and largely a matter of debate.

The Proposed Conservation Paradigm

We believe that recovery under the ESA should be defined as “attainment of the status at which a species’ persistence in nature in the foreseeable future is reasonably ensured.” We also believe and suggest that Congress had as its unstated, but characteristically anthropocentric intent, that the “foreseeable future” is logically delimited by the adult lifespan of a typical human (~6 decades). However, shorter timeframes may be more tenable under the ESA because outcomes of ecological processes can be meaningfully predicted for only much shorter periods. For example, based on the opinions

![Figure 1. Key characteristics that distinguish the Contemporary Recovery Paradigm from the Proposed Conservation Paradigm.](image-url)
of species’ experts, a recent ESA listing consideration for the westslope cutthroat trout (O. c. lewisi) defined foreseeable future as 2 to 3 decades (Federal Register 68: 46989 [7 August 2003]). Lack of a precise measure of “foreseeable future” notwithstanding, as embodied in our Proposed Conservation Paradigm (Figure 1), meeting our defined level for recovery should be treated as just the initial, albeit critical, step leading to long-term conservation of the species.

The Proposed Conservation Paradigm necessitates that the USFWS employ considerable scientific and technical expertise when making listing decisions under the ESA. Because management actions essential to achieving recovery must neutralize key threats to the listed species’ survival (see also Clark et al. 2002), it is vital that the USFWS make objective, scientifically-based determinations of those threats in the species’ listing rule, thereby providing clear guidance for development of the recovery plan. If those threats are not clearly specified, concerned entities should request—as part of the formal comment process for the proposed listing rule—that the USFWS clearly identify the threats and objectively substantiate their importance to the species’ risk of extinction. The ESA’s section 6(a) states, “In carrying out the program authorized by this Act, the Secretary [of the Interior] shall cooperate to the maximum extent practicable with the States.” (see also Interior 59: 34274 [1 July 1994]; Petersen 1999). We believe such cooperation includes the state’s review of listing petitions, meaningful participation in species’ status reviews, and peer review of subsequent draft findings that describe proposed listing decisions under the ESA.

Equally important, recovery criteria prescribed in the recovery plan approved by the USFWS should precisely “mirror” (sensu Doremus 2000) the criteria used to determine that listing of the species was warranted, as specified in ESA section 4(f)(1)(B)(ii) (see also Kline 2001). For example, if a decline in species’ abundance toward probable extinction in the “foreseeable future” was the key criterion used for listing, a stable or increasing trend in abundance for a comparable period—supported by a similar degree of statistical rigor—should be the key criterion for recovery. In making the listing determination, the USFWS should compile and analyze available information on the species and seek objective peer review of those analyses and related interpretations from the states and other qualified entities. Such analyses and reviews would need to be closely coordinated to be accomplished within the 12-month reporting schedule specified by the ESA. In our opinion, that rigid schedule, coupled with the near absence of external review prior to the publication of findings in the Federal Register, contributes to development of the subjective, vague criteria often used by the USFWS to support a “warranted” listing finding, whereas the effectively unlimited time available for recovery promotes recovery-team development of rigorous recovery criteria. We believe that important disparity in criteria results largely from actions that served to “move the [recovery] goalpost” and has led to much of the uncertainty or ill feeling evident to us in cutthroat trout recovery programs today.

While we believe these elements of the Proposed Conservation Paradigm should be applied to future listing determinations, the disparity between listing and recovery criteria that we have described for the currently listed cutthroat trout would remain. One way to remove the present impasse and move the recovery process forward would be for the USFWS to conduct a comprehensive status review, as required by the ESA’s section 4(c)(2), to determine whether a species should remain listed, be assigned to a different listing status, or be delisted. That review should compare the recent trend in overall abundance and distribution of the listed species, assess ongoing efforts to protect and conserve the species, evaluate the efficacy of extant regulatory mechanisms apart from the ESA (Doremus 2000; Doremus and Pagel 2001), and contrast other relevant information with that which was known at the time of listing. The review’s conclusions should be published in the Federal Register and pertinent comments should be received and considered by the USFWS. Although the ESA requires such reviews of the status of each listed species at least once every five years, they have seldom been performed and not at all for the listed cutthroat trout. Status reviews of listed species may also be initiated by concerned entities acting through the delisting-petition process specified in the ESA’s section 4(b)(3)(A).

The heart of the Proposed Conservation Paradigm is the long-term conservation of the species that follows ESA-mandated recovery (Figure 1). The USFWS has acknowledged the practical reality that the states bear much of the responsibility for managing federally-listed species (USFWS 1990). Likewise, the states have principal responsibility for managing the extant inland cutthroat trout not listed under the ESA, and there are numerous state and federal laws and regulations that serve to protect the fish and their habitats. Presently, each of the unlisted cutthroat trout subspecies is being managed under a long-term conservation program. These subspecies have range-wide or state-level, interagency conservation agreements (e.g., westslope cutthroat trout, Bonneville cutthroat trout [O. c. utah]), conservation plans (e.g., Colorado River cutthroat trout [O. c. pleuriticus], Yellowstone cutthroat trout [O. c. bouvieri], Rio Grande cutthroat trout [O. c. virginalis]), and conservation-implementation teams, all entirely apart from the ESA. Some plans have been revised on the basis of contemporary ecological concepts or improved knowledge of the subspecies and each has a record of positive performance. For some subspecies, information on present status, the progress of ongoing conservation actions (e.g., Hepworth et al. 2002), and other pertinent conservation concerns is routinely compiled. May (2002) reported that the five unlisted cutthroat trout subspecies mentioned above benefited from 780 conservation actions implemented between 1999 and
2002 and valued at more than $19 million. The amounts per subspecies were several times larger than for the listed cutthroat trout. Efforts are also underway to post this information at appropriate web sites. In our opinion, making such information readily available to everyone is especially important to demonstrating conservation success as well as assuaging concerns that the states and their cooperators are unable to effectively conserve the cutthroat trout without the aid of the ESA (e.g., Petersen 1999; Doremus 2000).

Despite the 32-year history of the ESA, no fish has been delisted as the result of recovery. In our opinion, that record is disappointing and may result, perhaps to a large degree, from common application of the Contemporary Recovery Paradigm, as we have described for the listed cutthroat trout. We also believe that application of the Proposed Conservation Paradigm, which clearly distinguishes between recovery under the ESA and subsequent long-term conservation of cutthroat trout, provides a more-collaborative approach for dealing with the real ecological, political, and social challenges of implementing this important law. Moreover, and perhaps more importantly, we believe the proposed paradigm provides for the long-term conservation of the cutthroat trout.

References


Analysis of Customs Trade Data to Characterize Importation of Live Bait

Introduction

So far, the economic growth forum has addressed the macroeconomic context of fish conservation (Czech and Pister 2005), microeconomic assumptions of economists (Krall 2005), an example of an important fisheries (Pacific salmon) declining in the face of a growing collection of economic sectors (Lackey 2005), and the economic link to invasive species introductions (Ericson 2005). We help illustrate the conflict between economic growth and fish conservation from a different angle, identifying a growing microeconomic sector that increasingly affects virtually all native fish species and fisheries.

The use of live bait by anglers, especially live bait fish, has resulted in a significant number of introductions in the aquatic environment throughout the United States and parts of Canada (Carlton 1992). Live bait organisms are commonly shipped from the point of collection or production to retail outlets. Studies addressing this transport of bait have primarily examined interstate trade (LoVullo and Stauffer 1993; Litvak and Mandrak 1993; Ludwig and Leitch 1996). However, there is substantial international trade in live organisms, some of which may be destined for international trade in live organisms, which greatly contributed to the collapse of the lake trout (Salvelinus namaycush) and whitefish (Coregonus spp.) fisheries in the Great Lakes. We started to think about international trade of live bait as a vector for aquatic invasive species introductions when a marine polychaete (Namalycastis sp.), native to Vietnam, starting arriving in the United States in the mid-1990s. The species is used for live fishing bait in the Chesapeake Bay region, where they are marketed as “nuclear worms.” Namalycastis sp. is one of the largest species of polychaete worms, growing to 3 m in length. The species inhabits low salinity, high water mud flats of the Mekong Delta, but is probably tolerant of a wide range of salinities. The species can tolerate high temperatures and low oxygen conditions (C. Glasby, Australia Museum of the Northern Territory, pers. comm.).

States have the authority to regulate what is imported, sold, and used as live bait. Few states prohibit importation of all live bait species. Many states list certain species as illegal to import, sell, and use as live bait. Federal regulations only cover imported earthworms used for fishing bait under the authority of the Federal Plant Pest Act. Shipments of live aquatic organisms imported into the United States are not well regulated and products are transported in many small shipments with diverse and changing sources and markets. It is thus difficult to assess what is coming in for a given pathway and to characterize the risk that organism poses to the aquatic environment.

To better characterize the risk of introducing aquatic invasive species to the aquatic environment through importation of live bait, our objectives were to: (1) estimate the quantity of live bait imported into the United States by source region, (2) identify spatial and temporal patterns of importation and distribution, and (3) clarify the taxonomy of imported bait.

Customs Data and Port Sampling

U.S. Customs Service (Customs) trade data from 1998–2000 were examined to characterize the trends in importation of live aquatic organisms. Import data are recorded by Harmonized Tariff Codes (HTC), many of which include but are not limited to live aquatic organisms. HTCs known to be imported for use as live bait are Canadian nightcrawlers (Lumbricus terrestris) that are primarily harvested in Canada, and a marine polychaete (Namalycastis sp.) that is imported from Vietnam.

Live bait can be introduced into the aquatic environment by accidental spillage, escape from the hook, or intentional dumping by the angler. Imported live organisms present three potential sources of invasion: (1) the target species that is intentionally imported, (2) morphologically similar species that are mistakenly collected with the target species, and (3) macro- and microscopic organisms that may be associated with the target organism, its gut contents, or packing materials. Ecological impacts of invasive species can include a significant loss of biodiversity, as well as primary threat to the integrity of various ecosystems’ structures and functions. Economically, the direct and indirect costs of invasive species are difficult to estimate. Even controlling a single unwanted invader can cost millions of dollars. For example, the United States and Canada are spending $14 million a year just to control the sea lamprey (Petromyzon marinus), which greatly contributed to the collapse of the lake trout (Salvelinus namaycush) and whitefish (Coregonus spp.) fisheries in the Great Lakes.
thought to include live bait ("Aquatic Invertebrates," "Bait, Other than Worms," "Live Fish," and "Live Worms") were reviewed to determine spatial and temporal patterns of live bait importation.

Customs trade data is useful in describing general trends in importation of live aquatic species; however, inferences are limited by: (1) taxonomic uncertainty associated with the database, (2) inability to separate live from non-living for some HTCs, (3) use of monetary value to describe shipment size rather than a biologically meaningful metric such as weight or number of organisms. Due to the limitations of Customs data, we decided to observe and sample shipments of live bait at ports of entry to help clarify taxonomy of imported bait, determine size of shipments, and to obtain more detailed records of origin and destination of shipments.

We met in 2000 with representatives from the U.S. Fish and Wildlife Service Law Enforcement (USFWS LE), Customs, and the U.S. Department of Agriculture (USDA), Animal and Plant Health Inspection Service to coordinate sampling and observation of live bait shipments at different ports of entry. At these meetings, it was determined that USFWS LE did not have sampling authority but that USDA could obtain samples. Before sampling occurred in 2001, however, USDA questioned their authority to obtain samples of live bait other than worms. They determined that authority does exist under the Federal Plant Pest Act to obtain samples of worms imported into the United States, but not to sample other live bait organisms unless they contain soil or plant matter that have the potential to become plant or animal pests in the country.

We initially chose seven ports to collect samples based on the volume and value of the shipments for the four HTCs in 2000 (Table 1). Customs suggested that we start off sampling in 2001 with two ports and for a two-month duration. We chose to sample at the ports of San Francisco, California, and Buffalo, New York, from 1 August–30 September.

Sampling criteria were provided to Customs, USFWS, and USDA inspectors at both ports (Table 2). Shipments arriving under any of the four HTCs and meeting specifications in the criteria would be flagged by Customs. Customs would then contact a USFWS inspector on duty. The inspector would visually inspect the shipment and then call us to determine whether or not USDA should take a sample. Being limited to taking only samples of worm shipments, we requested that USFWS inspectors obtain documents (declarations, invoices) that could provide information about taxonomic identification, destination, and origin of shipments from the other tariff codes. If there was a shipment in which we wanted to obtain a sample, permission from the importer to obtain a specimen would be sought or alternatively an attempt to purchase the organism from a retail outlet could be made.

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**Table 1. The most active ports of entry in 2000 for Harmonized Tariff Codes that may contain live bait.**
Live Worm Imports

The total value of "Live Worms" imported into the United States exceeded $70 million from 1998–2000. Organisms were imported from 10 countries, with the majority of imports originating from Canada, France, Netherlands, and the United Kingdom. The remaining shipments came from Belgium, Chile, Turkey, Australia, Vietnam, and Japan. Shipments from Canada likely consisted mainly of Canadian nightcrawlers. Canada is the only country exempt from obtaining an USDA permit to import live worms into the United States.

"Live Worm" imports entered the United States at 21 different ports of entry during 1998–2000, with Detroit, Michigan; Port Huron, Michigan; and Buffalo, New York accounting for the majority of activity. There was a pronounced seasonal trend in "Live Worm" import activity all three years with a peak in shipments occurring in June and July.

Aquatic Invertebrates Imports

Total customs value of "Aquatic Invertebrates" imported into the United States exceeded $4 million from 1998–2000, with the majority of imports coming from Canada, South Korea, and Australia. "Aquatic Invertebrates" entered the United States at 21 different ports of entry during 1998–2000, with Massena, New York, and Portland, Maine, accounting for the majority of the activity. There was no pronounced seasonal pattern to import activity of "Aquatic Invertebrates" during all three years, which may indicate that most of the shipments were intended for food consumption or the aquarium industry, and not for use as live bait. Customs data for "Aquatic Invertebrates" encompasses a broad category that could include an array of invertebrate taxa imported either live or fresh.

Live Fish Imports

Total customs value of "Live Fish" imported into the United States exceeded $3 million from 1998–2000, with the majority of imports coming from Canada, Mexico, Taiwan, and China. "Live Fish" entered the United States at 22 different ports of entry during 1998–2000, with San Francisco, California; Calais, Maine; and Los Angeles, California accounting for the majority of the activity. There was a seasonal trend in import activity of "Live Fish" shipments during all three years with a slight peak in imports from April–June.

Bait, Other Than Worm Imports

Total customs value of "Bait, Other Than Worms" imported into the United States was approximately $500,000 from 1998–2000, with imports coming from Canada and South Africa. "Bait, Other Than Worms" entered the United States at Detroit, Michigan; Port Huron, Michigan; Buffalo, New York; and Memphis, Tennessee during 1998–2000. There was a seasonal trend in import activity of "Bait, Other Than Worms" shipments during all three years with a peak in imports from May–July.

Modes of Transportation

"Live Worm" imports were shipped into the United States from 1998–2000 primarily by truck (92%), followed by air (6%), and vessel (2%). "Aquatic Invertebrate" imports arrived in the United States primarily by truck (59%), followed by air (26%), and vessel (15%). "Live Fish" imports arrived by truck (56%) and air (44%). "Bait, Other Than Worms" imports arrived primarily by truck (99%) and 1% arrived by air. For all Harmonized Tariff Codes, a greater proportion of shipments arrive into the United States by trucks because of the large number of shipments originating in Canada.

Port Sampling Results

Sampling was curtailed by the events of 11 September, but samples from the Port of San...
Francisco were obtained in August. Shipments of nuclear worms arrived at that time, as did a shipment of European nightcrawlers (*Dendrobaena veneta*) from France. No calls from inspectors at the Port of Buffalo occurred, likely due to the fact that the worms that came through the port at the time were from Canada.

**Discussion**

If implemented on a longer time scale and at more ports, port sampling could significantly contribute to our understanding of what organisms are coming in under these four HTCs and which ones are destined for the live bait trade. Due to workload constraints of each of the three agencies, this approach is not practical.

By examining USFWS declarations, USDA earthworm permits, and Customs consigne information we are slowly piecing together what kind of organisms are coming in under the four HTCs. USDA conducts risk assessments on terrestrial earthworms proposed for import and require permits for all countries except Canada. Customs consignee information, which we obtained for 2001 data, reveals the company that exports the shipments.

Examination of consignee information for the "Live Worms" HTC for 2001 revealed that the majority of imports are Canadian nightcrawlers imported from Canada and European nightcrawlers imported from the Netherlands and France. These species are imported for use as live bait, but because they are terrestrial in origin they pose more of a risk to terrestrial resources. The greatest threat to the aquatic environment is posed by importation of marine bait, which included bloodworms (*Glycera dibanchiata*) and sandworms (*Nereis virens*) imported from Europe, and polychaete worms imported from Vietnam (*Namalycasis* sp.) and Korea (*Perinereis aibuhitensis*). Examination of consignee for the "Bait, Other Than Worms" HTC revealed that many of the importers are fly fishing companies, which may indicate that artificial bait such as lures and flies are being imported from Canada and South Africa.

An Internet search for each importer was conducted, resulting in identification of only one web site, for a wholesale bait company located in Wisconsin. The company sells 13 different species of minnows and shiners, some of which are cultured in ponds on site or ponds in Louisiana or Arkansas. Some of the other species are harvested from wild populations occurring in rivers and streams in the Great Lakes region. The emerald shiner (*Notropis atherinoides*) offered for sale is harvested from wild populations in Wisconsin, Michigan, Ohio, New York, Ontario, and Quebec, and may account for the company's import activity. Based on cursory examination of consignee information, the majority of the imports coming in under the "Live Fish" and "Live Aquatic Invertebrates" HTC are destined for the seafood and aquarium trade.

**Conclusion**

Economists often use the phrase *ceteris paribus*, which is Latin for "all else being equal." *Ceteris paribus*, the impact of the live bait trade on fish conservation and ecological integrity will continue to grow as a function of economic growth. The bait trade is one in a collection of economic sectors that grow as an integrated whole. Some of the most clearly linked sectors include fishing tackle, boats, and outboard motors. Furthermore, labor (a major factor of economic production along with land and capital) grows as a function of population and, *ceteris paribus*, more people equates to more fishing and more live bait trade. We conclude that the live bait trade is one manifestation of a larger challenge to fish conservation, i.e., economic growth, and that fish conservation will ultimately require macroeconomic policy reform in addition to regulating individual sectors.

**Literature Cited**


